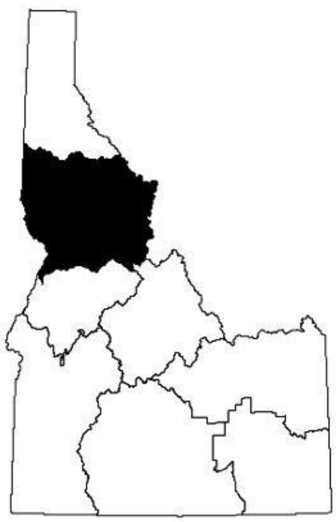
FISHERY MANAGEMENT INVESTIGATIONS

# IDAHO DEPARTMENT OF FISH AND GAME
## FISHERY MANAGEMENT ANNUAL REPORT

**Ed Schriever, Director**



## CLEARWATER REGION
## 2016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## TABLE OF CONTENTS

## LIST OF TABLES

## LIST OF FIGURES

## DEER CREEK RESRVOIR: TIGER TROUT EVALUATION


## ABSTRACT

Beginning in 2014, tiger trout (Brown Trout *Salmo trutta* X Brook Trout *Salvelinus fontinalis)* have been stocked into Deer Creek Reservoir (DCR) annually. The purpose of these stockings was to introduce a fish that would prey upon an introduced (illegally or through birds) population of Golden Shiner *Notemigonus crysoleucas* and provide desired fishing opportunities. In 2016, we shifted from stocking tiger trout as fingerlings, to stocking them at catchable-size to improve their survival, increase predation on Golden Shiners, and increase angler utilization. The 2016 fish survey resulted in the capture of 163 tiger trout, compared to none captured in the 2015 survey. Golden Shiners were present in the stomach contents of 52% of the tiger trout, showing that Golden Shiners are being utilized by tiger trout. This is also supported by the large reduction in Golden Shiner catch-per-unit-effort from fall gill net sampling in 2015 and 2016. Angler exploitation of tiger trout was estimated at 7.3% in 2016, much lower than the 15.1% exploitation estimated for Rainbow Trout *Oncorhynchus mykiss* in DCR during 2012. While the estimated exploitation rate for tiger trout is quite low, this is likely due to the fact this was the first year of stocking these fish at catchable sizes, and anglers are not yet fully aware of their presence. We recommend conducting a more intensive monitoring program during 2017 to provide more information on the Golden Shiner population. Additionally, we recommend evaluating angler utilization again in 2017.



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## INTRODUCTION

Deer Creek Reservoir (DCR) was constructed during 2003. It was created by damming Deer Creek, a tributary of Reeds Creek that flows into Dworshak Reservoir. DCR is an important part of the region's lowland lake program as it provides a location for trout harvest in an area where all stream fishing is under restrictive harvest regulations (two trout per day). It also adds diversity to our fishery program as it is the only lowland lake managed only as a cold-water fishery. DCR was originally managed to provide a put-and-take Rainbow Trout (RBT) *Oncorhynchus mykiss* fishery, and put-and-grow Westslope Cutthroat Trout (WCT) *O. clarkii lewisi* and Brook Trout (BKT) *Salvelinus fontinalis* fisheries.

Golden Shiner (GS) *Notemigonus crysoleucas* became established in DCR soon after the reservoir filled. We are unsure of the means of establishment as the department did not stock GS into DCR. Upon their discovery in 2006, the reservoir was subsequently treated with rotenone in 2006 and 2010 in attempts to eradicate this invasive species (Hand 2010; Hand et. al. 2013). The logic supporting attempts to eradicate GS from DCR were: 1) GS are effective planktivores, and an overabundance of GS in DCR could potentially reduce the quality and quantity of the zooplankton available for trout; and 2) the possibility that GS might spread downstream into Dworshak Reservoir which supports an important kokanee fishery that has been found to generate more than $4 million in fishing-related expenditures annually for the surrounding communities (IDFG, *unpublished data*).

Unfortunately, GS were again sampled in DCR in 2012. The failure of the rotenone treatments was likely due to a combination of factors including the high level of habitat complexity within the reservoir (large slash piles had been left to provide habitat), springs/seeps that could provide clean water refuge, and GS resistance to rotenone. Golden Shiners have a natural resistance to rotenone and are capable of developing a higher resistance to rotenone which would increase each time the same population is exposed (Orciari 1979). If the initial renovation was not 100% effective, any surviving GS would have the potential of creating a rotenone-resistant population. Additionally, GS were found in several ponds in the nearby Schmidt Creek drainage (near Weippe, Idaho) during the construction of Deyo Reservoir. Nez Perce Tribe fisheries biologists also reported finding GS in nearby drainages including Orofino Creek and Jim Ford Creek. This indicated that GS were widespread locally and complete eradication would be nearly impossible.

With the realization that rotenone treatments were not effective at eliminating GS, we assessed the potential of stocking tiger trout (TT; Brown Trout *Salmo trutta* X Brook Trout). Tiger trout have been reported to be a more effective predator than either parent species (Sheerer et. al. 1987). Our hope was that the TT would utilize GS as a prey source, thereby providing a unique, new fishing opportunity in the region. Additionally, the TT could potentially provide some control of GS abundance, thus improving the food base for trout that depend on zooplankton. With this in mind, we began stocking fingerling TT (50 - 75 mm) in DCR in the spring of 2014.

Surveys conducted in 2014 confirmed our concern regarding zooplankton abundance and quality (i.e. size), as sampling revealed a substantial decline in zooplankton sizes and abundance compared to previous data when GS were not present (Hand et al. 2017). This decline in food resources may have been a primary reason why only one TT was sampled in 2014 and 2015, and would likely result in future decreased growth and survival of trout dependent on this food source. Golden Shiners were present in the stomach contents of only Rainbow Trout (RBT) *Oncorhynchus mykiss* and Brook Trout (BKT) over 250 mm. The apparent lack of success in establishing a TT population through the stocking of fingerlings suggested that changes to our stocking strategy were necessary. Decreasing or eliminating the stocking of fingerling trout, and stocking larger trout (TT, RBT, and/or BKT >250 mm) could increase their likelihood of survival, increase predation of GS, and decrease the predation pressure on zooplankton. Additional sampling in 2015 again resulted in the capture of few TT, confirming our concerns that fingerling stockings were not successful. Thus, we changed our strategy and began stocking approximately 2,500 “catchable size” TT (170 - 360 mm) annually in June 2016. Sampling has been conducted annually since then to continue evaluating the effectiveness of TT predation on GS, and angler exploitation of the catchable size TT.

## OBJECTIVES

1. Evaluate whether tiger trout stocked at catchables sizes (~250 mm) can effectively prey upon Golden Shiners.

2. Evaluate angler exploitation of tiger trout.

## STUDY AREA

Deer Creek Reservoir is located in Clearwater County, Idaho, 21 km north of Pierce, Idaho (Figure 1). It is a 47.0-ha reservoir located at an elevation of 1,006 m. It has a maximum depth of 11 m, and a maximum volume of 936,000 $m^3$. Completed in 2003, it is the second-newest reservoir in the state of Idaho. It was created by damming Deer Creek, a tributary of Reeds Creek that flows into Dworshak Reservoir. The watershed is owned by Potlatch Corporation and is used primarily for timber harvest. Idaho Department of Fish and Game leases the reservoir property from Potlatch Corporation. Today, the reservoir is used extensively by boaters and anglers and provides unique trout fishing opportunities.

## METHODS

Electrofishing and gill net surveys were conducted on October 27, 2016, to evaluate the trout populations, and determine stomach contents of fish collected. Boat-mounted electrofishing was performed using pulsed D.C. current from a Honda EU7000iAT1 generator and a Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. One hour of electrofishing was divided into six 10-minute subsamples, with fish sampled in each subsample processed and recorded separately. This allows for the calculation of variance estimates necessary for comparisons to other surveys and for calculating the appropriate sample size for future surveys (IDFG 2012). Electrofishing was conducted along the shoreline in a clockwise direction, with each subsample started at the locations marked in Figure 2. The survey was conducted at night, and we attempted to net all fish observed. Species, total length (mm), and weight (g) were recorded for each fish sampled.

Gill net sampling consisted of four nets set for 12 hours overnight. Gill net sites were non-randomly spread throughout the reservoir as much as possible, while being set in areas least likely to be damaged by woody debris (Figure 3). The presence of large quantities of wood throughout the reservoir, especially the upper 1/3, prevent us from placing nets in many locations. Floating monofilament experimental gill nets 36-m long and 1.8-m high were used. The nets were divided into six equal size panels with bar mesh sizes of 10.0, 12.5, 18.5, 25.0, 33.0 and 38.0 mm. Monofilament diameter ranged from 0.15 to 0.20 mm. Nets were set perpendicular to the shoreline and anchored in place to prevent the net from drifting. The smallest mesh end was tied to shore, and the largest mesh end towards the middle of the reservoir. Weight (g) and total length (mm) was recorded for all trout sampled. Only total lengths (mm) were recorded for GS.

Trout that could not be released alive from the gill nets were dissected to identify stomach contents. This analysis was conducted at the reservoir as fish were removed from the gill nets. Stomach contents were separated into four different categories: empty, GS, insects, and detritus (no other items were observed). Items were recorded as presence/absence.

Catch-per-unit-effort (CPUE; ± 90% confidence intervals) was calculated as an index of abundance to compare with previous years. Significant differences in CPUE between years were determined to be those where 90% confidence intervals do not overlap. Mean length of fish (± 90% confidence intervals) were compared by species between years using a standard two-sample *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.05$. Data analysis does not include comparisons to data collected in 2015 because surveys were conducted at a different time of year (fall vs. spring).

Angler exploitation was estimated for hatchery catchable-size TT (170 - 360 mm) stocked on June 22, 2016. Tiger trout ($n = 99$) >190 mm were tagged at the reservoir as they were stocked. These fish were randomly selected by netting fish directly from the hatchery truck into a holding tank. Fish below the minimum tagging size were measured for total length (mm) and released into the reservoir. Fish were tagged with Hallprint model FD-94 anchor tags. Each fish tagged was measured for total length (mm) and weight (g). Tagging data (date, location, species, length, weight, tag number) was submitted to IDFG Nampa Research Office and uploaded to the IDFG "Tag You're It" database. Tagging, data entry, and calculation of angler exploitation rates (based on reported tags) were conducted based on methodology of the IDFG "Tag You're It"/Fish Database program (Meyer et. al 2010).

## RESULTS

The electrofishing and gill net surveys resulted in the catch of 393 fish, including RBT ($n = 187$), TT ($n = 163$), BKT ($n = 34$), GS ($n = 7$), and WCT ($n = 2$). Rainbow Trout ranged in length from 226 to 376 mm with an average (90% C.I.) of 316 mm (± 3; Figure 4). This average length was significantly larger ($\alpha = 0.05$; $P < 0.001$) than the 270 mm (± 4) average in 2014. Tiger trout ranged in length from 170 to 427 mm and averaged 288 mm (± 7; Figure 5). No TT were sampled in 2015. Brook Trout ranged in length from 179 to 358 mm and averaged 285 mm (±15; Figure 6). Golden Shiners ranged in length from 101 to 193 mm and averaged 139 mm (±26; Figure 7). This average length was significantly larger ($\alpha = 0.05$; $P < 0.001$) than the average observed in 2014 (95 mm; ±12). Two WCT, 237 and 295 mm in length, were sampled. The CPUE for gill nets in 2016 (38.3 fish/net) was lower than in 2014 (140.5 fish/net; Figure 8). Stomach contents were analyzed for 50 RBT, 25 TT, and 8 BKT collected from the gill nets. Aquatic insects were the most common item identified in BKT (63%) and RBT (38%), whereas GS were the most common item identified in TT (52%; Figure 9). Golden Shiners were also found in 13% of BKT and 10% of RBT. For all three trout species, no GS were present in fish < 270 mm (Figure 9).

Tiger trout tagged on June 22, 2016 ($n = 99$) ranged in length from 195 to 346 mm (Figure 11). Six fish (6%) between 158 to 195 mm were not tagged due to small size (Figure 11). During a one-year period, anglers reported only three tags from TT they harvested (June and July 2016; May 2017). No tagged TT were reported as having been caught and released. The angler exploitation (fish harvested) rate through 365 days at large was 7.3% (Table 1). The angler total use (fish harvested plus fish released) rate through 365 days at large was also 7.3% (Table 1). The three tags returned were from fish tagged at 225, 285, and 303 mm in length.

## DISCUSSION

In contrast to previous sampling where no TT were caught, 163 TT were sampled in 2016. This indicates that stocking these fish at “catchable size (160 - 370 mm)” instead of as fingerlings greatly improved their survival. Comparing the length distribution of TT stocked vs. those sampled in surveys does not show any fish sampled in sizes smaller than those stocked. This, in addition to only one TT being sampled in 2014 - 2015, further indicating that stocking TT as fingerlings was not successful. Rainbow Trout catch rates and average size has increased as well. Recent reductions in the number of RBT stocked into DCR is likely contributing to this increase in average size, as fewer fish will improve the food resources available for all fish in the reservoir (Figure 12). Other influencing factors may include fewer RBT are being caught/harvested, or that anglers are shifting effort towards TT. This is something we need to monitor over the next few years. If effort/harvest does shift away from RBT, we will need to consider modifying our stocking strategy. Brook Trout catch rates have remained similar during the last three years of sampling, while average length increased in 2016. While we have maintained consistent stocking rates of 2,500 - 3,500 fingerlings per year since 2014, the reduction in overall trout stocking rates (primarily from reducing RBT) should improve food resources and growth rates. Golden Shiner average length has increased each year of sampling, while CPUE has fluctuated. The increase in average length is likely due to a maturing population, as more GS reach larger sizes. However, some of this could also be due to predation of smaller GS by trout. We recommend conducting more intensive sampling for GS during summer peak months to better analyze this population.

Although stomach content analysis indicates that TT are successfully preying upon GS, no GS were found in any trout <270 mm in length. This indicates that trout, regardless of species, don’t begin preying upon GS until they reach this size. This is important to note, as 33% of the TT stocked in 2016 were <270 mm in length. Therefore, these fish will likely need a year in the reservoir before they are large enough to begin preying on GS. The presence of GS in 52% of TT stomachs, much lower GS gill net CPUE, and a size shift towards only larger GS (Figure 1-8), suggests that TT are likely having an impact on the GS population. However, based on sampling conducted in 2014, GS catch rates peaked in July, then dropped steadily through November for all sample methods (gill nets, electrofishing, minnow traps; Hand et al. 2017). This suggests that GS are less susceptible to our sample gear as the water cools down in the fall. In order to more thoroughly evaluate this program, we recommend a more intensive monitoring program for 2017 and standardizing sampling dates for GS across years to make appropriate comparisons. This should include sampling for GS in the summer when they are most active, monthly zooplankton sampling, and fall trout sampling.

Estimated angler exploitation and total use rates through 365 days-at-large (both 7.3%) were less than half the rates of 15.1% and 17.3% estimated for RBT in DCR in 2012 (Table 1). However, this is based on only three total tag returns, so conclusions about exploitation should be made with caution. These exploitation rates may be biased slightly high due to tagging fish only >190 mm. These smaller fish would likely be caught at a lower rate than larger fish. While the exploitation rate estimate for TT are quite low, this was the first year of stocking these fish at catchable sizes. As such, there are not as many of them in the reservoir as there are RBT, and not all anglers are aware of this new opportunity and may not be targeting them yet. Once anglers “discover” TT, catch and harvest rates for this species will likely increase. We recommend evaluating exploitation again in 2017.

## MANAGEMENT RECOMMENDATIONS

1. Repeat surveys in fall 2017 using electrofishing and gill netting to assess potential changes in trout populations.

2. Sample Golden Shiner in summer to capture peak population for comparison with previous samples.

3. Continue to stock tiger trout at lengths >250 mm.

4. Monitor catch and harvest of Rainbow Trout and tiger trout in 2017 to detect changes in angler preference or effort.

Table 1. Angler exploitation of tiger trout stocked into Deer Creek Reservoir, Idaho, in 2016, based on angler-reported, t-bar anchor tags through 365 days at large.

| | | Disposition | | | Adjusted exploitation | | Adjusted total use | |
|---|---|---|---|---|---|---|---|---|
| Tagging date | Tags released | Harvested | Harvested b/c tagged | Released | Estimate | 90% C.I. | Estimate | 90% C.I. |
| 6/22/2016 | 99 | 3 | 0 | 0 | 7.3% | 6.4% | 7.3% | 6.4% |

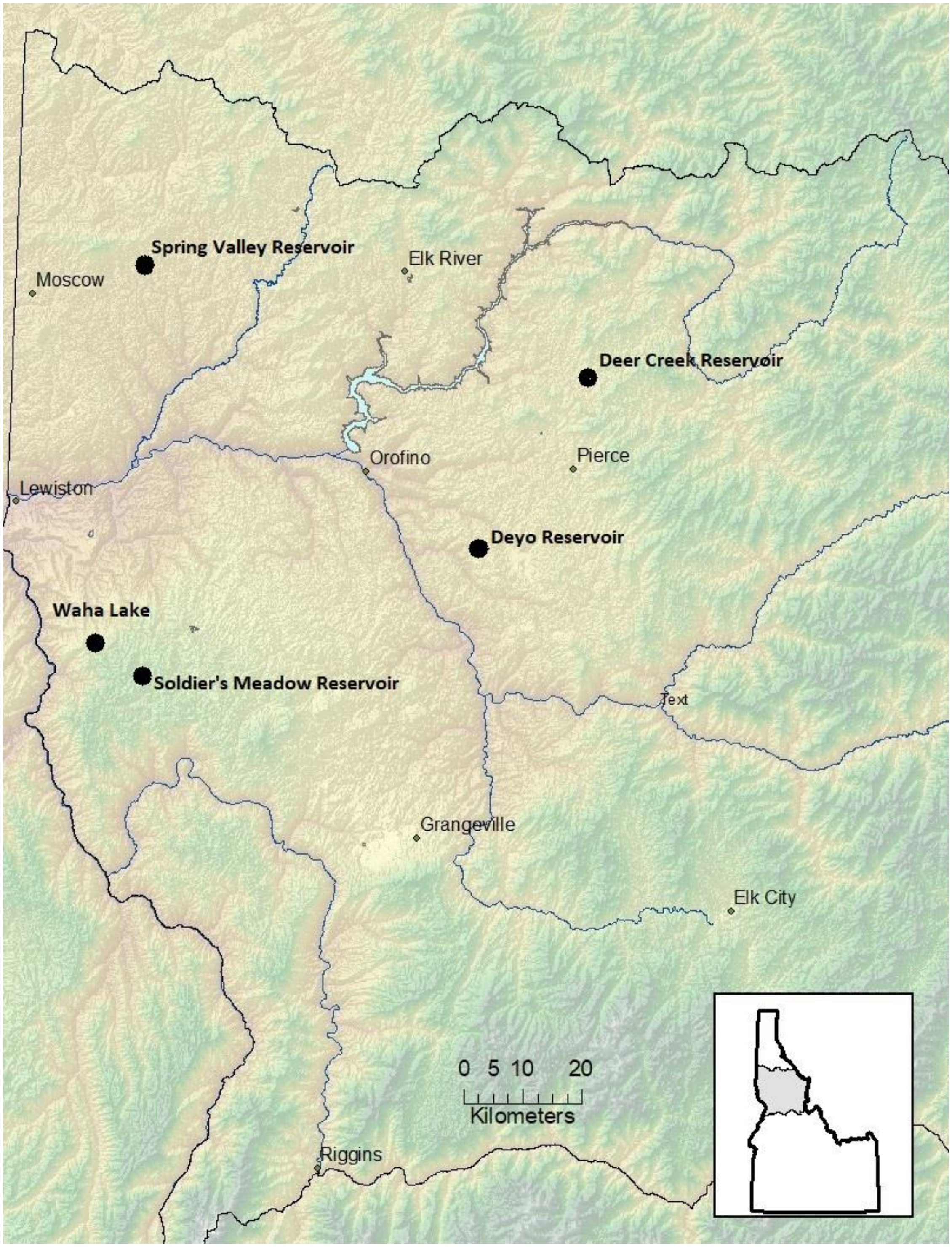


Figure 1. Map showing locations of reservoirs surveyed in the Clearwater Region, Idaho, during 2016.

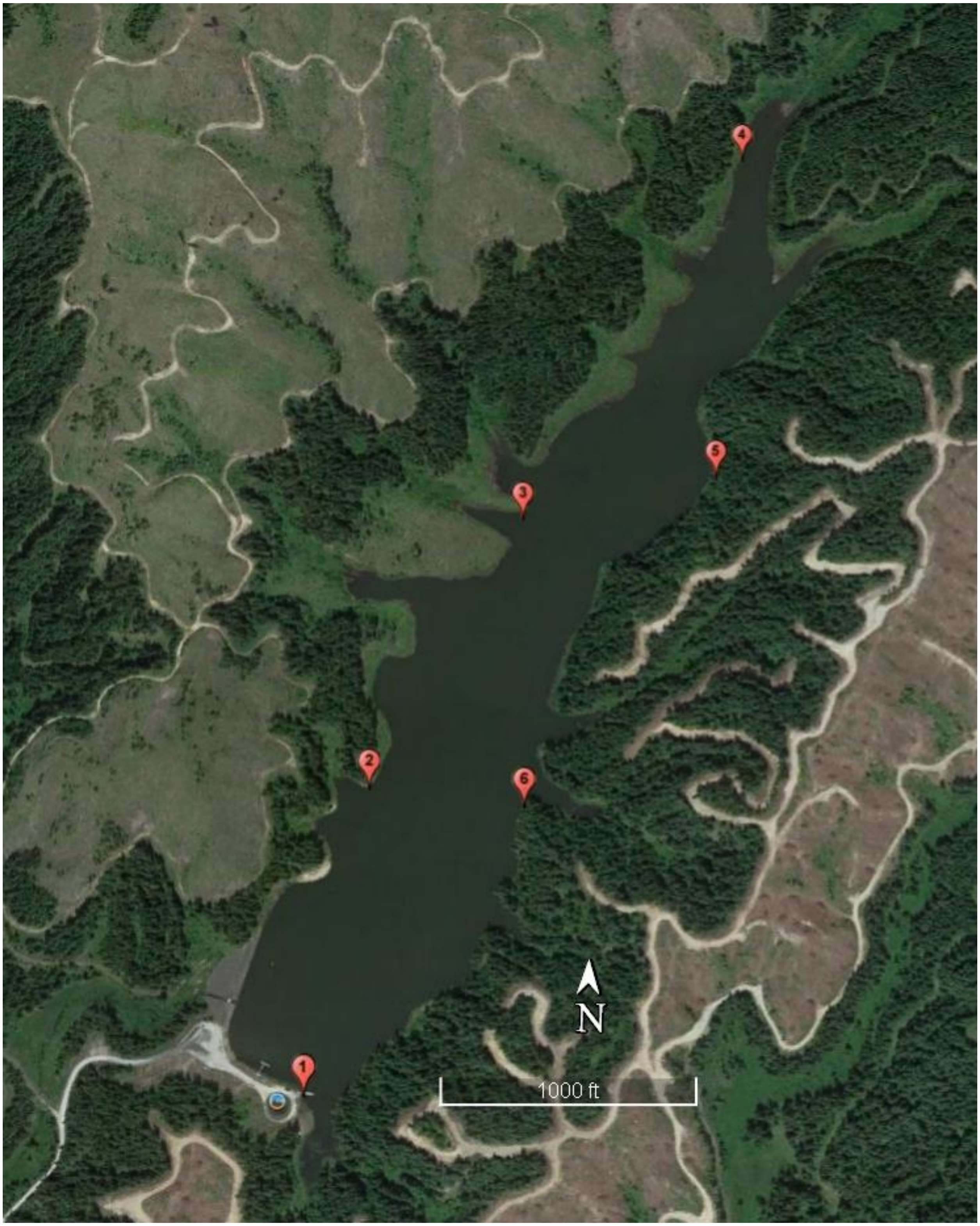


Figure 2. Locations of starting points for electrofishing survey sub-samples on Deer Creek Reservoir, Idaho, in 2016.


Figure 3. Location of four gill nets set in Deer Creek Reservoir, Idaho, on October 22, 2018.

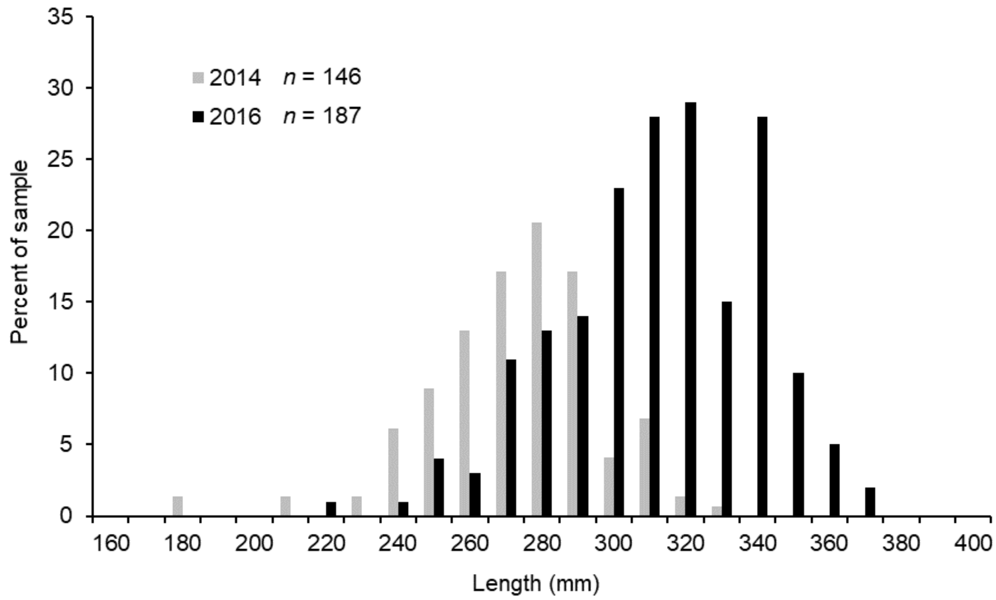


Figure 4. Length-frequency distribution of Rainbow Trout sampled from electrofishing and gill nets in Deer Creek Reservoir, Idaho, during late October-early November 2014 and 2016.

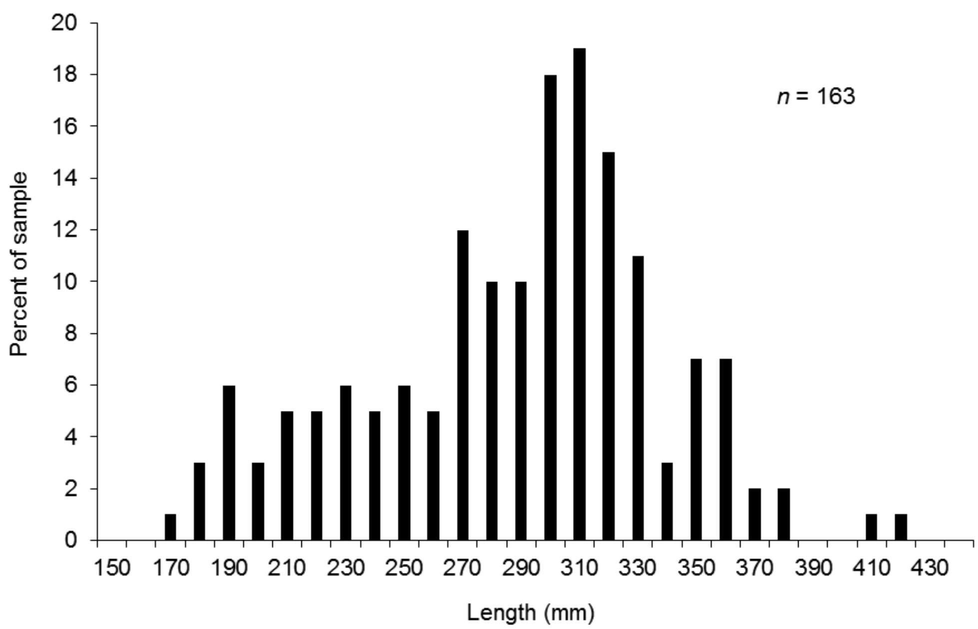


Figure 5. Length-frequency distribution of tiger trout sampled from electrofishing and gill nets in Deer Creek Reservoir, Idaho, during November, 2016.

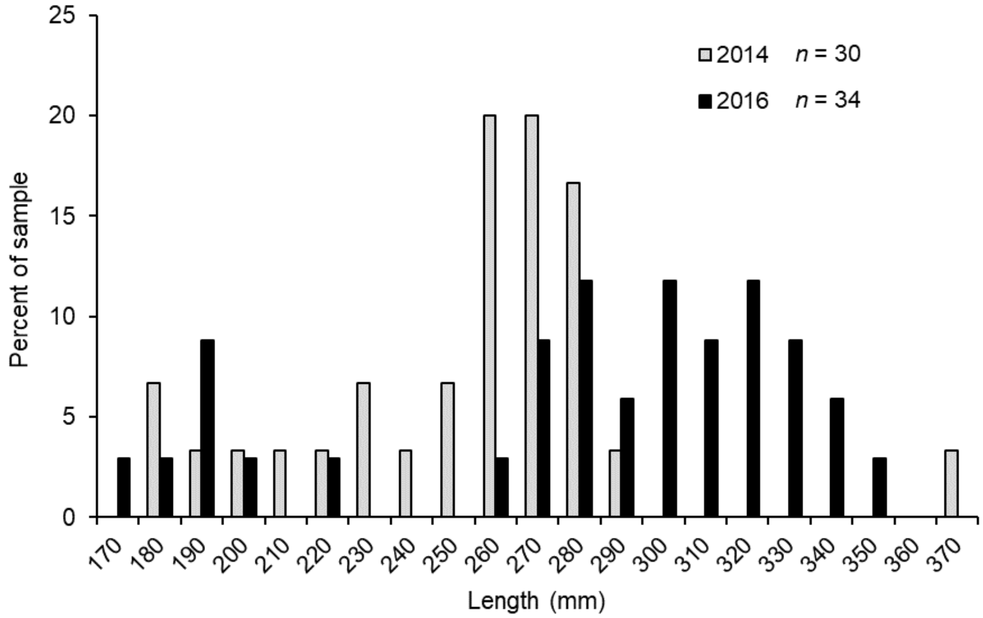


Figure 6. Length-frequency distribution of Brook Trout sampled from electrofishing and gill nets in Deer Creek Reservoir, Idaho, during late October-early November, 2014 and 2016.

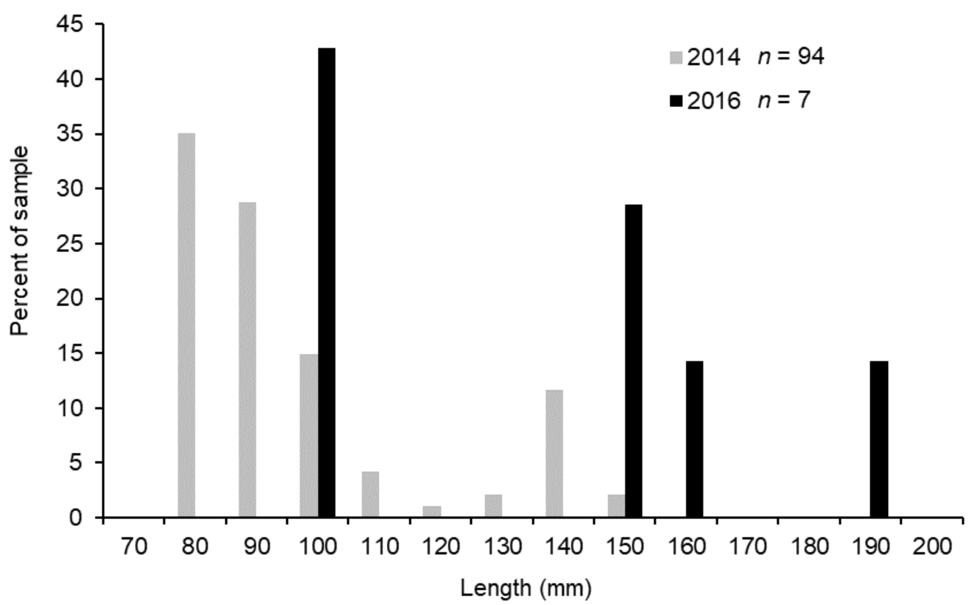


Figure 7. Length-frequency distribution of Golden Shiners sampled from electrofishing and gill nets in Deer Creek Reservoir, Idaho, during late October-early November, 2014 and 2016.

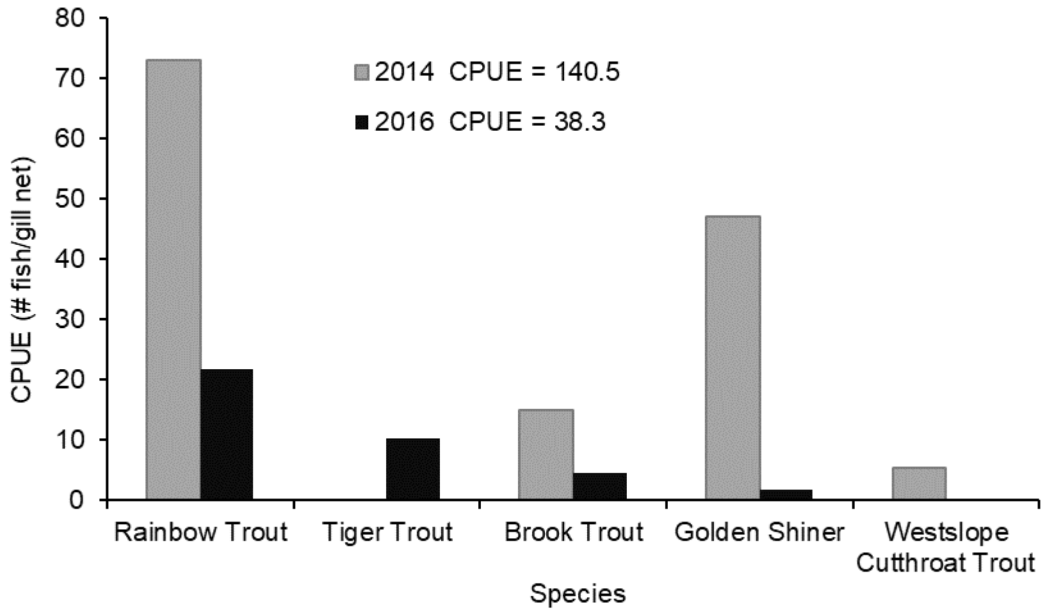


Figure 8. Catch-per-unit-effort (CPUE; fish/net) for fish sampled from gill nets in Deer Creek Reservoir during late October-early November, 2014 and 2016.

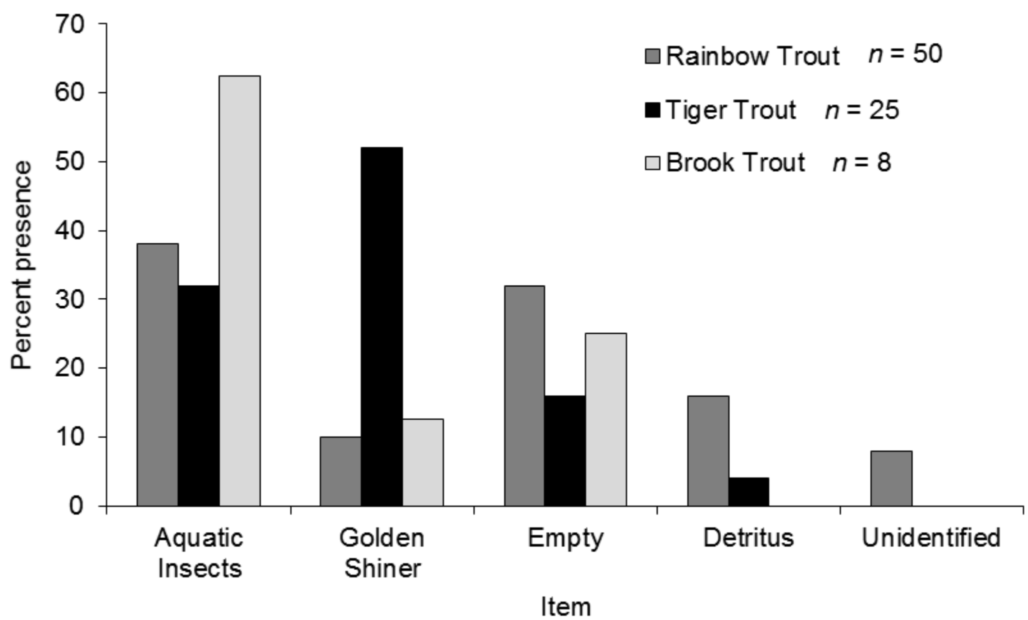


Figure 9. Contents of stomach samples from trout collected by gill nets in Deer Creek Reservoir, Idaho, on October 26, 2016.

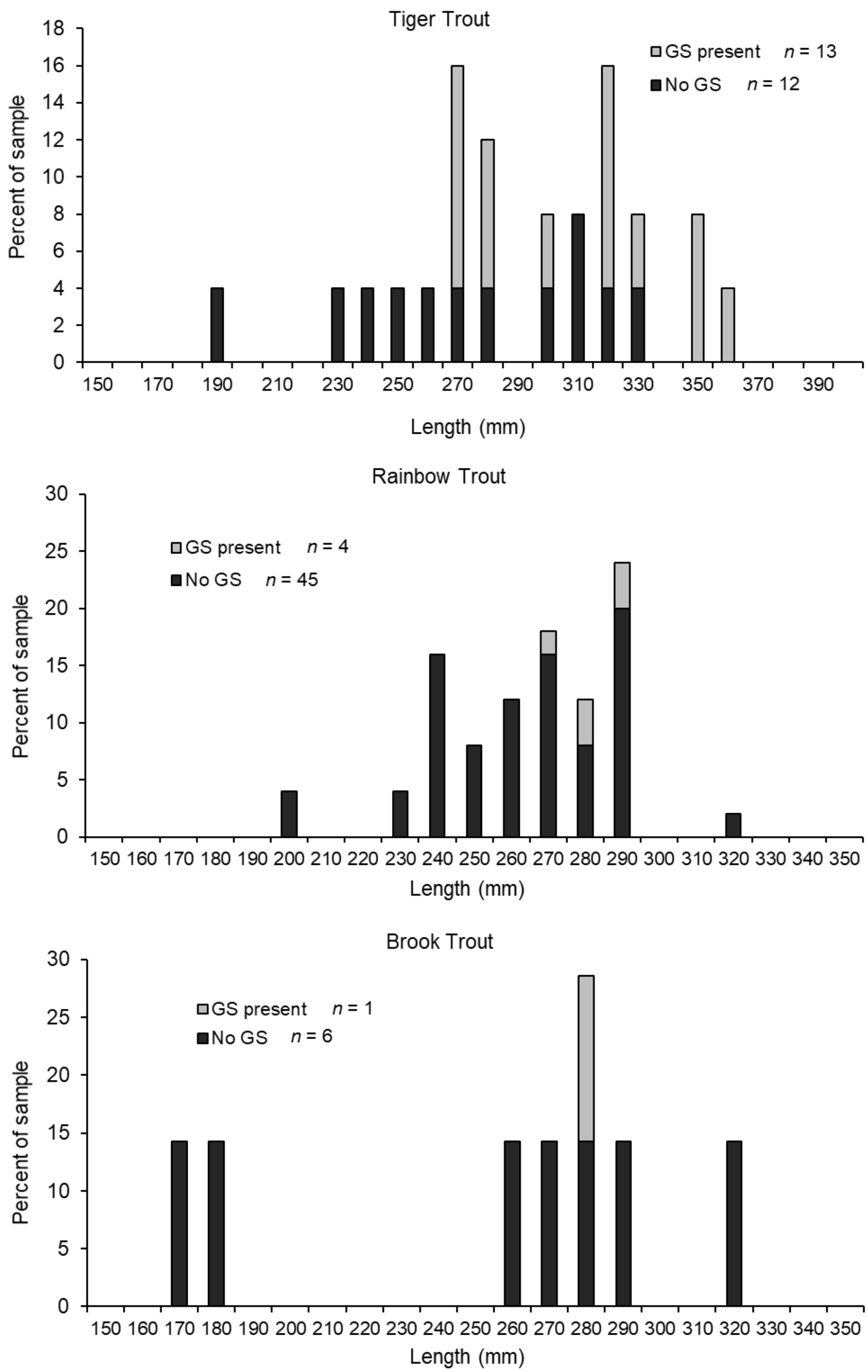


Figure 10. Length-frequency distributions of trout with GS present in stomach samples versus those without GS present in Deer Creek Reservoir, Idaho, for fish examined in 2016.

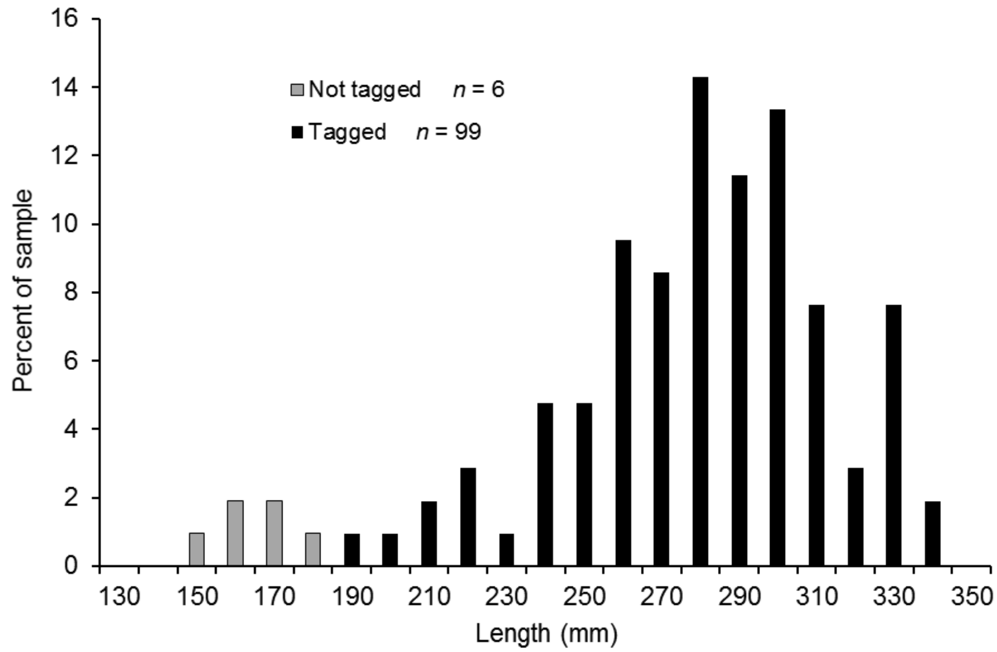


Figure 11. Length-frequency distribution of tiger trout tagged in Deer Creek Reservoir, Idaho, and those not tagged due to small size, in 2016.

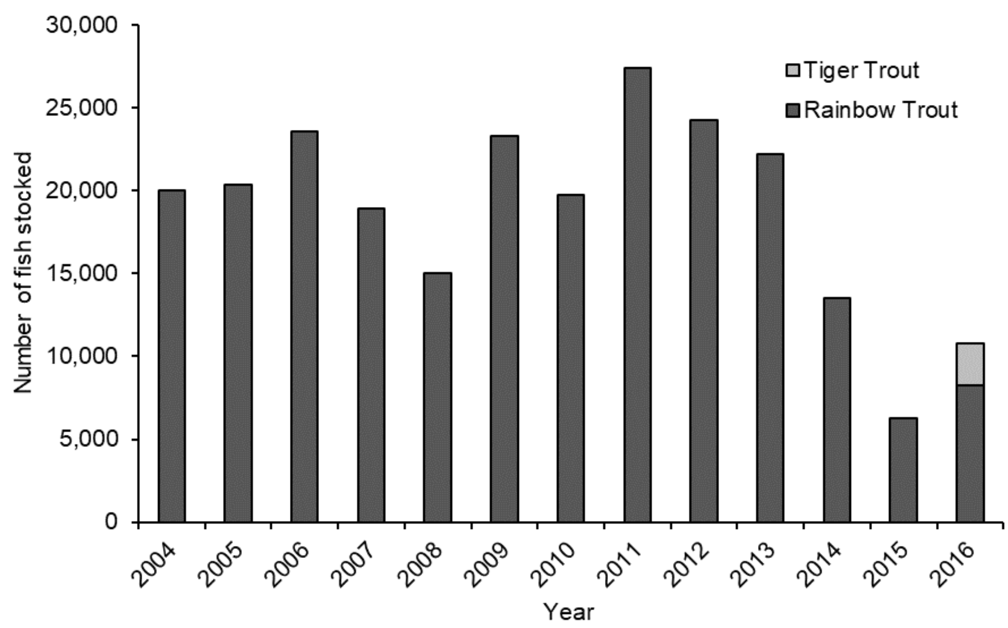


Figure 12. Number of catchable size fish (170 - 360 mm) stocked in Deer Creek Reservoir, Idaho, from 2004 to 2016.

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# DEYO RESERVOIR AND SCHMIDT CREEK EVALUATIONS


## ABSTRACT

Surveys were conducted in Deyo Reservoir in 2016 to provide baseline data for evaluating the effects of restrictive regulations for Largemouth Bass *Micropterus salmoides* (LMB; 406 mm minimum size, two fish bag limit) that were implemented in 2016. Both LMB and Bluegill (BG) *Lepomis macrochirus* populations have increased in average length and proportional size distribution over the past three years. However, with no LMB over age-4 in the population other than those originally stocked in 2012, there is no replacement occurring for harvested fish. To mitigate potential overharvest, 313 LMB were translocated into Deyo Reservoir from Bonner Lake and Smith Lake, near Bonner's Ferry, Idaho. These fish averaged 196 mm, mostly smaller than our 300 mm minimum target size. However, they are large enough to prey on BG and will help fill gaps in the size structure of the LMB population. Our data also suggests that preferred zooplankton (such as Daphnia) were in low abundance and of small size, indicating cropping by stocked Rainbow Trout (RBT) *Oncorhynchus mykiss* and the numerous small BG. The change to the LMB regulations and translocation of additional LMB should improve predation on BG and have a positive impact on the zooplankton population. At this time, we recommend maintaining the regulations to improve LMB predation on BG and reduce LMB harvest. We also recommend conducting a fish survey again in 2017 to continue these evaluations. Schmidt Creek was sampled April-November, 2011-2016, to determine if the construction of Deyo Reservoir could negatively impact steelhead. Stream flow, water temperature, and dissolved oxygen (DO) were monitored downstream of the reservoir. During the summer months of 2012 - 2016, Schmidt Creek (where steelhead occurred) maintained perennial flows. In fact, stream flow below the dam changed from intermittent to perennial after the reservoir was built, largely due to normal leakage/seepage that occurs through the dam. Our data suggest that summer base flow in Schmidt Creek has increased since the construction, providing more area for rearing steelhead in this critical, low-flow period. Maximum daily water temperatures in Schmidt Creek continued to be highly variable but remained well below lethal limits for *O. mykiss* during most of the year. It appears that the natural seepage through the dam has also positively affected water temperatures. This cooler water helps reduce the temperature in the lower stretches of the creek, which should benefit steelhead rearing habitat. Dissolved oxygen concentrations remained above 6 mg/L in the lower reaches of Schmidt Creek throughout the study. Although the water seeping through the dam showed DO levels <6.0 mg/L through much of the summer, DO levels recovered to >9.0 mg/L by the time water reached the lower reaches of Schmidt Creek. Thus, DO was not a concern in regards to water quality for fish downstream of the dam. Based on the information collected during this study, it appears that Deyo Reservoir is not having a negative effect on steelhead in Schmidt Creek. Therefore, we recommend ending the monitoring project at this time and continue with current dam operations.



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## INTRODUCTION

Deyo Reservoir was constructed in 2012 to provide a new recreational fishery and an economic boost to the local economy with minimal negative biological impacts (DuPont 2011). Prior to the construction of Deyo Reservoir, fish surveys in upstream reaches of Schmidt Creek in close proximity to the planned dam location observed Longnose Dace *Rhinichthys cataractae,* as the only native fish species in that area (DuPont 2011). Surveys conducted on Schmidt Creek by the Idaho Department of Environmental Quality in 2002, found juvenile steelhead *Oncorhynchus mykiss* and sculpin sp within 60 m of the mouth of Schmidt Creek. Given the presence of steelhead in lower Schmidt Creek, an agreement was made with the U.S. Fish and Wildlife Service to monitor stream flow and temperature in the area where steelhead were found to occur and assess whether Deyo Reservoir was negatively affecting these attributes (DuPont 2011). Monitoring would occur for five years after construction, and if deleterious effects could be related to the construction of the Deyo Reservoir, IDFG would modify water release strategies as needed.

The management strategy for Deyo Reservoir is to provide a “two-story” fishery, with both cold and warm-water species. This included annual stocking of catchable-size hatchery Rainbow Trout (RBT) *Oncorhynchus mykiss*, and introducing and managing for balanced and self-sustaining populations of Largemouth Bass (LMB) *Micropterus salmoides* and Bluegill (BG) *Lepomis macrochirus*. Largemouth Bass and BG were first introduced in 2012, when 100 LMB (255 - 403 mm) and 350 BG (98 - 177 mm) were collected from Winchester Lake and Spring Valley Reservoir and translocated into Deyo Reservoir. A follow-up electrofishing survey in 2015 sampled few LMB >300 mm, while BG were plentiful but averaged only 87 mm. Additionally, reports from anglers and the campground host at Deyo Reservoir indicated that LMB harvest was higher than anticipated. With no LMB over age-4 in the population other than those originally stocked, harvest may have a significant impact on larger size classes of fish until naturally-spawned fish were large enough to replace those harvested. In an effort to protect the LMB that were large enough to spawn and prey upon BG, restrictive regulations for LMB (daily limit of 2, none under 406 mm) were implemented in 2016. In addition, we determined additional LMB supplementation was needed. To evaluate the effectiveness of this rule change and supplementation, additional monitoring of Deyo Reservoir was necessary.

## OBJECTIVES

1. Establish baseline fishery data to evaluate the effectiveness of restrictive Largemouth Bass rules and supplementing the Largemouth Bass population with larger fish.

2. Supplement the Largemouth Bass population in Deyo Reservoir with larger size classes from other Idaho Lakes.

3. Assess zooplankton abundance and size as a food source for juvenile Largemouth Bass and Bluegill.

4. Monitor flow, temperature, and dissolved oxygen (DO) in Schmidt Creek to assess whether the construction of Deyo Reservoir is having deleterious effects on steelhead occurring downstream, and whether water release strategies need to be modified.

## STUDY AREA

Deyo Reservoir was constructed in 2012 by damming Schmidt Creek, a tributary to Lolo Creek, Idaho. The reservoir is located approximately 5 km west of Weippe, Idaho, at an elevation of 920 m (Figure 1). At full pool, the surface area of the reservoir is 22.3 ha. It has a maximum depth of approximately 10 m, a mean depth of approximately 5 m, and a maximum volume of approximately 678,000 $m^3$. The upper end of the reservoir has been developed into a wetland area to provide habitat for waterfowl and other wildlife. The drainage basin is composed of a mix of forest and cropland. Facilities at the reservoir include a campground with both full hookups and primitive sites, numerous fishing docks (including ADA accessible), boat ramp, picnic pavilion, and toilets.

Schmidt Creek contains designated critical habitat for steelhead from its mouth to 1.1 km upstream. The end of steelhead critical habitat is 2.7 km below the Deyo Reservoir Dam site. Habitat upstream of the reservoir is unsuitable for steelhead, Bull Trout, spring Chinook Salmon, or Coho Salmon spawning or rearing. Stream flow within Schmidt Creek is considered intermittent within the reservoir project area and potentially perennial in lower reaches depending on annual precipitation within the drainage area. Summer low flows are believed to limit steelhead abundance. The channel upstream of the reservoir site generally is dry during summer months. Schmidt Creek is dominated by silt substrate and the riparian area is dominated by grasses. The original streambed and surrounding area has been altered by logging, construction of the logging millpond, road construction, and stream channelization to improve grazing (DuPont 2011).

## METHODS

### Fish community survey

To evaluate the fish community in Deyo Reservoir, an electrofishing survey was conducted on May 17, 2016. Boat-mounted electrofishing was performed using pulsed D.C. current from a Honda EU7000iAT1 generator and a Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. The one hour of electrofishing was divided into six 10-minute subsamples, with fish sampled in each subsample processed and recorded separately. This allows for the calculation of variance estimates necessary for comparisons to other surveys and for calculating the appropriate sample size for future surveys (IDFG 2012). Electrofishing was conducted along the shoreline in a clockwise direction, with each subsample started at the locations marked in Figure 13. The survey was conducted at night, and we attempted to net all fish observed. Species, total length (mm), and weight (g) were recorded for each fish sampled. Catch-per-unit-effort (CPUE; fish/h) and weight-per-unit-effort (WPUE; kg/h) with associated ± 90% confidence intervals were calculated for total catch and each species to compare with previous years. Significant differences between years were determined to be those where 90% confidence intervals do not overlap. Mean length of fish (± 90% confidence intervals) were compared by species between years using a standard two-sample *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.1$.

Proportional Size Distribution (PSD; Guy et al. 2007; Neumann et al. 2012) and relative weights ($W_r$; Wege and Anderson 1978; Neumann et al. 2012) were calculated for LMB and BG. Proportional Size Distribution was calculated to provide information on population size structure using the following formula:

$$PSD = \frac{\# fish \geq qualitysize}{\# fish \geq stocksize} * 100$$

Quality size and stock size correspond to lengths considered to be the minimum size at which anglers will first catch the species (stock) and consider the fish to be of desirable size (quality). These lengths are 200 mm and 300 mm for LMB and 80 mm and 150 mm for BG (Gablehouse 1984; Neumann et al. 2012). Proportional Size Distribution values of 40 - 70 for LMB and 20 - 40 for BG are considered to be indicative of a balanced population (Anderson 1980).

Proportional Size Distribution decision models were developed to assess the current predator-prey dynamics in each reservoir (Schramm and Willis 2012). These models plot predator (LMB) PSD versus prey (BG) PSD. The PSD values for LMB and BG can each fall into three categories: low, desirable, or high. Thus, there are nine possible predator:prey PSD size structure scenarios. Explanations for each situation and recommended management actions are detailed in Schramm and Willis (2012).

Relative weight ($W_r$) was calculated for LMB and BG to provide information on their body condition:

$$W_r = \frac{W}{W_s} * 100$$

where $W$ is the observed weight of the fish and $W_s$ is the length-specific standard weight predicted by a weight-length regression. This equation is:

$$log_{10}\ Ws = a + (b * log_{10}\, total\ length)$$

where $a$ is the intercept and $b$ is the slope of standard weight equations developed for many fish species (Wege and Anderson 1978; Neumann et al. 2012). Relative weights were calculated for each LMB and BG measured and weighed, and a separate scatterplot of these relative weights was developed for each species. A trend line (linear regression) within this data was plotted for each species to depict how their relative fitness changed with size.

**Largemouth Bass translocation**

Largemouth Bass were collected from Bonner Lake and Smith Lake near Bonner's Ferry, Idaho, on June 16, 2016. Fish were collected using two boats equipped with the electofishing equipment described earlier in this document. Electrofishing was conducted at night along the shoreline until approximately 150 fish were collected from each reservoir. These fish were held in a hatchery truck until morning when they were translocated on June 17$^{th}$.

**Zooplankton**

Zooplankton samples were collected bi-weekly from August 19 - October 11, 2016. Samples were collected with a Wisconsin-style plankton net (80 micron mesh, 30 cm diameter mouth). The boat was anchored at the deepest location on each lake based upon bathymetric maps and depth finder readings. When anchoring the boat, the anchor was slowly dropped and slack in the anchor line was let out to let the boat drift away from the anchor location. Three vertical tows were taken from that location. Tows were started 1.0 m above the bottom of the lake to avoid disturbing sediment. Depth of tow was recorded on each sample jar. Samples were rinsed into sample jars and stored in 70% ethyl alcohol. A Rite-in-the-Rain label was placed inside the sample jar. Samples were labeled with date, reservoir, number of tows, depth of tow, and personnel present.

Laboratory analysis was conducted based on a protocol developed previously for regional mountain lake surveys (Hand et al. 2016b). Zooplankton samples were diluted into a known volume container (typically 100 ml) and 5-ml aliquots were then subsampled. Subsamples were counted until 200 of the most dominant families were observed. The density of zooplankton in each individual tow was then estimated expanding the subsample estimate by total volume to the tow. Tow volume ($\pi$) was calculated by:

$$\pi \cdot r^2 \; x \; h$$

where $r$ = radius of the net and $h$ = depth of tow.

Zooplankton was counted based on three phylogenetic orders: Cladocera, Cyclopoida, and Calanoida. Within Cladocera (most common zooplankton), we identified individuals down to one of the following: Family Chydoridae, *Daphnia sp.*, *Ceriodaphnia sp.*, or *Bosmina sp.* In addition, the first 30 individuals of each category per sample were measured under the dissecting microscope to establish a length distribution for the sample.

### Schmidt Creek monitoring

Schmidt Creek was monitored April - November, 2011 - 2016 for stream flow ($m^3/s$), temperature (°C), and dissolved oxygen (DO; mg/L) at a monitoring location (Lower Station) approximately 10 m upstream from its confluence with Lolo Creek (N46.355800°, W-116.052637°; Figure 14). Temperature was recorded hourly using a HOBO™ temperature logger. Dissolved oxygen was recorded bi-weekly using a YSI model 550A meter. Stream flow was recorded bi-weekly using an OTT MF Pro flow meter. Flow rate was calculated using OTT's Mid-Section method, which uses velocity (taken at mid-depth) at multiple sites on a cross sectional transect of the stream (OTT Hydromet 2018). During 2016, a second monitoring station (Upper Station) was established 20 m downstream of Deyo Reservoir dam (N46.367655°, W-116.015615°) to monitor differences in these parameters with the Lower Station (Figure 14).

## RESULTS

### Fish community survey

The electrofishing survey resulted in the capture of 809 BG and 56 LMB. The catch rates for each of the six 10-minute samples ranged from 96 to 215 fish/sample, with a mean CPUE (90% C.I.) of 865 fish/h (± 184; Figure 15). The CPUE was the lowest since sampling began in 2014, although it was not statistically different from any other sample year (Figure 15). The CPUE for LMB (56 fish/h; ± 25) increased from the 29 fish/h (± 13) observed in 2015, but was not significantly different from previous years (Figure 16). The CPUE for BG (809 fish/h; ± 169) was lower than the 1,331 fish/h (± 557) observed in 2015, but was also not significant different from previous years (Figure 16). The mean WPUE (90% C.I.) of 28.6 kg/h (± 6.7) was slightly higher than 2015, but was not significantly different (Figure 17). The WPUE for LMB was 12.8 kg/h (± 5.4), higher than the 5.0 kg/h (± 3.8) in 2015, but not significantly different (Figure 18). The WPUE for BG was 15.8 kg/h (± 3.1), lower than the 21.2 kg/h (± 10.2) in 2015, but not significantly different (Figure 18).

Largemouth Bass sampled ranged from 130 to 410 mm in length, with an average total length (90% C.I.) of 214 mm (± 20; Figure 19). Average length was not statistically different from the 200 mm (± 26) average observed in 2015 ($\alpha = 0.05$; $P = 0.2361$). Fifteen of the 56 fish sampled (26.8%) were >300 mm in length. Largemouth Bass proportional size distribution (PSD) was 79, higher than the 31 calculated in 2015 (Figure 20). Relative weights ranged from 66 to 127 (average of 89) and tended to increase as the size of the fish increased (Figure 21).

Bluegill sampled ranged from 71 to 194 mm in length, with an average of 99 mm (± 2; Figure 22). Average length was significantly larger than the 87 mm (±0.6) average observed in 2015 ($\alpha = 0.05$; $P = <0.0001$). Most of the fish (90%) sampled were between 90 - 119 mm total length. Bluegill PSD was 1.4 in 2016, slightly higher than the 0.1 calculated in 2015 (Figure 20). Relative weights ranged from 54 to 145, with an average of 99 (Figure 23). Relative weight was similar across the range of lengths, but lower than the 135 calculated in 2015.

**Largemouth Bass translocation**

A total of 313 LMB were collected from Bonner Lake and Smith Lake near Bonner's Ferry, Idaho, on June 16, 2016. These fish ranged from 124 to 488 mm in length, and averaged 196 mm (±4; Figure 24). They were subsequently translocated into Deyo Reservoir on June 17, 2016, to supplement the LMB population in the reservoir.

**Zooplankton**

Bosmina were the most abundant zooplankton taxa, averaging 12,090/$m^3$ (Figure 25). Bosmina was also the most abundant taxa in August - October sampling in 2013 and 2014 (Hand et al. 2016b; Hand et al. 2017). However, densities in 2016 were lower than the average density of 36,741/$m^3$ for sampling conducted since 2013. Daphnia densities averaged 6,035/$m^3$, similar to the average seen in sampling since 2013 (6,677/$m^3$). Cyclopoida (5,114/$m^3$) were the next most abundant taxa (Figure 25). This density was very similar to the average of 5,166/$m^3$ seen in sampling since 2013. Calanoida, Ceriodaphnia, and Chydoridae were also present in low numbers (<1,000/$m^3$). The average length of Daphnia collected ranged from 0.37 to 0.60 mm, similar to samples collected in 2013 and 2014 (Figure 26).

**Schmidt Creek monitoring**

During the course of this project, stream flow at the Lower Station generally ranged from an April high of approximately 0.05 $m^3/s$ to summer lows of <0.01 $m^3/s$ in July - October (Figure 27). Flows typically declined slowly from April into mid-July after spring run-off, then increased again in late October with fall precipitation. However, flows in 2012 were substantially higher in the spring, reaching 0.31 $m^3/s$ in May (Figure 27). Flows at the Upper Station ranged from a high of 0.03 $m^3/s$ in April, to a low of 0.2 cfs in late August (Figure 28). During 2016, flows were higher at the Lower Station than at the Upper Station throughout sampling (Figure 28). No de-watering of the stream channel was observed during this project at either sample site.

Water temperatures measured at the Lower Station were highly variable, but were generally below 15°C until early July, and above 15°C from then until early September (Figure 29). However, 2011 was an exception to this trend, as water temperatures never exceeded 15°C. This may be due to cooler average air temperatures experienced in June - August, 2011, compared to 2012 - 2016, based on climate data for Pierce, ID (U.S. Climate Data 2019). Maximum daily water temperature exceeded 20.0°C for a four days in 2012, six days in 2015, and sixteen days in 2016. The peak high temperature recorded was 21.8°C on July 29th, 2016. In 2016, average daily water temperature at the Upper Station was 13.8°C (Figure 30). This station recorded 15 days with a maximum temperature >20.0°C, with a peak of 24.6°C on May 7th. Average daily water temperature at the Lower Station was also 13.8°C, with 16 days with a maximum temperature >20.0°C (Figure 30). Water temperatures were higher at the Upper Station until early June, and again starting in early October.

Average monthly DO levels at the Lower Station ranged from 6.2 to 16.1 mg/L, and averaged 10.1 mg/L over the course of this study (Figure 31). Dissolved oxygen levels generally were highest in the spring and fall, and lowest during the summer months. The exception to this trend was 2011, when DO was higher during the summer. This was due to the cooler water temperatures recorded in 2011 compared to 2012 - 2016. As discussed above, this was likely a result of the cooler air temperatures that occurred in 2011. In 2016, average monthly DO measured at the Upper Station was 4.8 mg/L, and ranged from a low of 1.0 mg/L in September to a high of 11.0 mg/L in June (Figure 32). These levels were substantially below what was measured at the Lower Station, which averaged 8.9 mg/L and ranged from a low of 6.2 mg/L in July to a high of 10.7 mg/L in November (Figure 32). The DO levels at the Lower Station were fairly stable throughout 2016. In contrast, DO levels at the Upper Station declined continuously from April - September.

## DISCUSSION

Restrictive regulations on LMB (406-mm minimum size limit, with a two-fish bag limit) were implemented on Deyo Reservoir in spring 2016. Information from anglers and the campground host, in addition to data from previous surveys, indicated a need to reduce harvest of LMB and improve the predator:prey balance. Minimum length limits are recommended for fish populations that exhibit low rates of recruitment and natural mortality, good growth rates, and high fishing mortality (Novinger 1984; Wilde 1997). They are generally used to protect the reproductive potential of fish populations, prevent overexploitation, increase angler catch rates, and promote predation on prey species (Noble and Jones 1993; Maceina et al. 1998; Iserman and Paukert 2010). For Deyo Reservoir, the objectives of this regulation change was to reduce harvest of LMB to improve the population size structure, and increase predation on BG.

The LMB population in Deyo Reservoir has been increasing in average length over the past three years (Figure 19). Additionally, PSD values have also increased since 2014 (Figure 20). In 2016, the LMB population also showed increases in the number of larger fish (>300 mm) and in CPUE. However, with no LMB <130 mm sampled in 2016, recruitment appears to be very low (Figure 19). In fact, the length-frequency distribution shows the lack of recruitment over the last few years, as the size of the smallest fish sampled has increased from 50 mm in 2014 to 130 mm in 2016 (Figure 19). While the improvements in size distribution is encouraging, the lack of LMB >300 mm is concerning. Angler harvest is likely having a negative impact on the population. With only approximately 125 LMB >300 mm stocked in 2012, even low harvest levels would have a large impact on the population. A creel survey of Deyo Reservoir estimated that 311 LMB were caught and 21 harvested in 2014. Assuming that all of the fish harvested were the larger fish stocked in 2012, approximately 17% of those fish were estimated to have been harvested in 2014 alone. This harvest rate of larger fish likely occurred in 2012, 2013, and 2015 as well. Although this exploitation rate is within range of angler exploitation rates of 8% - 35% calculated for several regional reservoirs (Hand et al. 2016a), with no fish over age-4 two in the population other than those originally stocked in 2012, there is no replacement occurring for those harvested. This will result in harvest having a larger impact than normal until naturally-spawned fish recruit to the fishery.

The BG population in Deyo Reservoir is also dominated by small fish, as evidenced by the PSD value of 1.4 calculated for the 2016 sample (Figure 20). With only three years of natural reproduction, we would expect PSD values to be low, as it will take several more years to have a fully-developed population. However, with so many smaller fish and few predators, there is concern that BG could overpopulate. With only 437 BG estimated to have been harvested in 2014 (assuming this is similar to the annual average harvest), there is not enough harvest to help reduce the population (Hand et al. 2017). The large numbers of small BG in the reservoir has likely made successful spawning and recruitment of LMB nearly impossible due to egg predation and competition for food resources (Anderson and Weithman 1978; Guy and Willis 1991). This is common in small impoundments when predator:prey dynamics are out of balance with few predators to control the overcrowding of prey species (Aday and Graeb 2012).

In addition to the fishing regulation changes, 313 LMB were translocated into Deyo Reservoir to increase the number of LMB in the reservoir. These LMB ranged from 124 - 488 mm in length, and averaged 196 mm (Figure 24). With 86.6% between 150 - 249 mm in length, these fish were mostly smaller than the 300 mm minimum target length. However, they were larger on average than the population in Deyo Reservoir, and are large enough to prey on BG. Additionally, they will help fill gaps in the size structure of the LMB population in Deyo Reservoir. These management strategies should improve LMB size structure and result in more large fish capable of preying upon BG. This, in turn, should reduce BG densities, resulting in improved growth rates and PSD. In the future, we should Floy-tag all LMB translocated into the reservoir to help evaluate angler exploitation of these fish. Additionally, we should conduct a mark-recapture of both species to provide more information on population size and help assess the impact of harvest.

Larger-sized zooplankton taxa, especially Daphnia, often compose a substantial portion of the diet of lake dwelling trout and juvenile warm-water species (Galbraith 1967; Hyatt 1980; Eggers 1982; Schneidervin and Hubert 1987; Aday and Graeb 2010). However, Daphnia, the most common taxa in Deyo Reservoir, averaged 0.37 - 0.60 mm in length (Figure 26). These average lengths are similar to those seen in previous years, but are substantially below the length (≥1.0 mm) preferred by BG at the sizes (>80 mm) seen Deyo Reservoir (Figure 22; Mittleback 1981). The data collected over the last few years suggests that while zooplankton were numerous, larger preferred zooplankton individuals (such as Daphnia) were in low abundance, indicating that they are likely being cropped off by stocked RBT and the numerous small BG (Mills et al. 1987; DeVries and Stein 1992). As BG size increases, their prey size increases, and they tend to select more towards macroinvertebrates, insects, and fish (Olson et al 2003; Spotte 2007). The changes to the LMB regulations and stocking additional larger LMB to improve predation on BG should have a positive impact on the zooplankton population. Improvements in predator:prey ratios following the introduction of additional predators has been shown to increase mean zooplankton size (Mills et al. 1987). We should continue to evaluate zooplankton over the next few years as we monitor the impacts of the LMB stocking and regulation changes on the fish and zooplankton community.

At this time, we recommend maintaining the restrictive regulations of a 406-mm minimum size limit with a two-fish bag limit for LMB. This should allow LMB to reach sizes where they can effectively prey on BG and allow for several years of spawning before both they can be legally harvested. We also recommend conducting a fish survey again in 2017 to evaluate the LMB population and potential changes in the predator:prey balance of the reservoir. If predator:prey balance has not improved, stocking more LMB >300 mm could be considered to improve predation on BG and improve reproductive success as larger fish are harvested. An angler survey should be considered to evaluate if restrictive regulations are impacting angler harvest rates of LMB and if BG are reaching a desirable size for harvest.

During the summer months of 2012 - 2016, Schmidt Creek (where steelhead occurred) maintained perennial flows. Although flows did not decrease to less than 0.01 $m^3/s$ in August in some years, we do not attribute this to the construction of Deyo Reservoir. In fact, based on our observations, stream flow below the Dam changed from intermittent to perennial after the reservoir was built. This is largely due to normal leakage/seepage that occurs through the dam. This flow has been measured to be approximately 0.003 $m^3/s$, which was captured by our measurements at the Upper Station in 2016 (Figure 28). This information suggests that, if anything, summer low flows in Schmidt Creek has increased since the construction of Deyo Reservoir providing more area for rearing steelhead in this critical low-flow period. It should be noted that flows in 2012 were highly unusual. This was caused by additional water being released from the reservoir during May and June to lower the water level so that a leak in the dam could be found and repaired. This repair occurred during the summer/fall, and the reservoir was refilled by natural runoff over the winter.

Maximum daily water temperatures in lower Schmidt Creek where steelhead occur seldom exceeded 20˚C. Studies have shown steelhead avoid temperatures in the mid 20˚C range, but temperatures at or near 20˚C are not detrimental, especially for short periods of time (Neilsen et al. 1994; Matthews and Berg 1997). The year when the warmest temperatures were observed was in 2016. However, in 2016, water temperature below the dam dropped considerably in early June. This was the period of time when flow over the spillway ended (from warmer lake surface water) and all that remained was seepage through the dam. Based on previous temperature profiles of Deyo Reservoir, the water seeping from the dam is coming from cooler, mid-level water around 3-m deep (Hand et al. 2016b). This water, typically around 14˚C matches up with the average summer water temperatures of around 14˚C at the Upper Station. This cooler water potentially helps reduce the temperature of water in the lower stretches of the creek, which should benefit steelhead rearing habitat.

Dissolved oxygen levels <6.0 mg/L are considered stressful to salmonids (Baldwin and Polacek 2002). However, DO concentrations in lower Schmidt Creek remained above 6 mg/L throughout the monitoring season each year since monitoring began in 2011 (Figure 31). In 2016, DO levels at the Upper Station declined throughout the summer, dropping below 6.0 mg/L in late June. This correlated with the lower DO level water seeping through dam. As DO levels in the reservoir decline during the summer, DO levels at the Upper Station declined as well (Figure 32). Although this water was low in DO for much of the sampling period, DO levels recovered to >9.0 mg/L by the time it reached the lower reaches of Schmidt Creek (Figure 32). This recovery is due to contact with air, and aeration of the water as it moves through high gradient areas. Thus, DO was not a concern in regards to water quality for fish downstream of the dam.

Based on the information collected during this study, it appears that Deyo Reservoir is not influencing stream conditions in Schmidt Creek in a manner where it will negatively impact steelhead that occur there. In fact, the increased flow of cooler water seeping through the dam throughout the summer is likely improving summer rearing conditions for steelhead that occur downstream of the reservoir. Therefore, we recommend ending the monitoring project at this time and continue with dam operations as have occurred over the last five years.

## MANAGEMENT RECOMMENDATIONS

1. Continue conducting fishery surveys to evaluate the influence of implementing more restrictive Largemouth Bass rules and supplementing the Largemouth Bass population in Deyo Reservoir.

2. Conduct a creel survey to evaluate the efficacy of the new Largemouth Bass rules (two fish limit, none <406 mm).

3. Collect basic age structure data for Bluegill and Largemouth Bass to estimate growth and mortality rates.

4. Floy-tag future translocations of Largemouth Bass to evaluate angler exploitation.

5. We recommend concluding the monitoring project as scheduled, and continue with current dam operations.

Table 2. Number of fish collected in each 10-minute sample, and catch-per-unit-effort (CPUE; fish/h) with 90% confidence intervals (CI) for an electrofishing survey of Deyo Reservoir, Idaho, in 2016.

| | Count of fish collected | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Species | Sample 1 | Sample 2 | Sample 3 | Sample 4 | Sample 5 | Sample 6 | CPUE (fish/h) | 90% CI |
| Largemouth Bass | 12 | 8 | 3 | 9 | 4 | 20 | 56 | 25 |
| Bluegill | 102 | 101 | 93 | 159 | 159 | 195 | 809 | 169 |
| Total | 114 | 109 | 96 | 168 | 163 | 215 | 865 | 184 |


Figure 13. Locations of starting points for electrofishing survey sub-samples on Deyo Reservoir, Idaho, in 2016.


Figure 14. Locations of Schmidt Creek monitoring stations near Deyo Reservoir, Idaho.

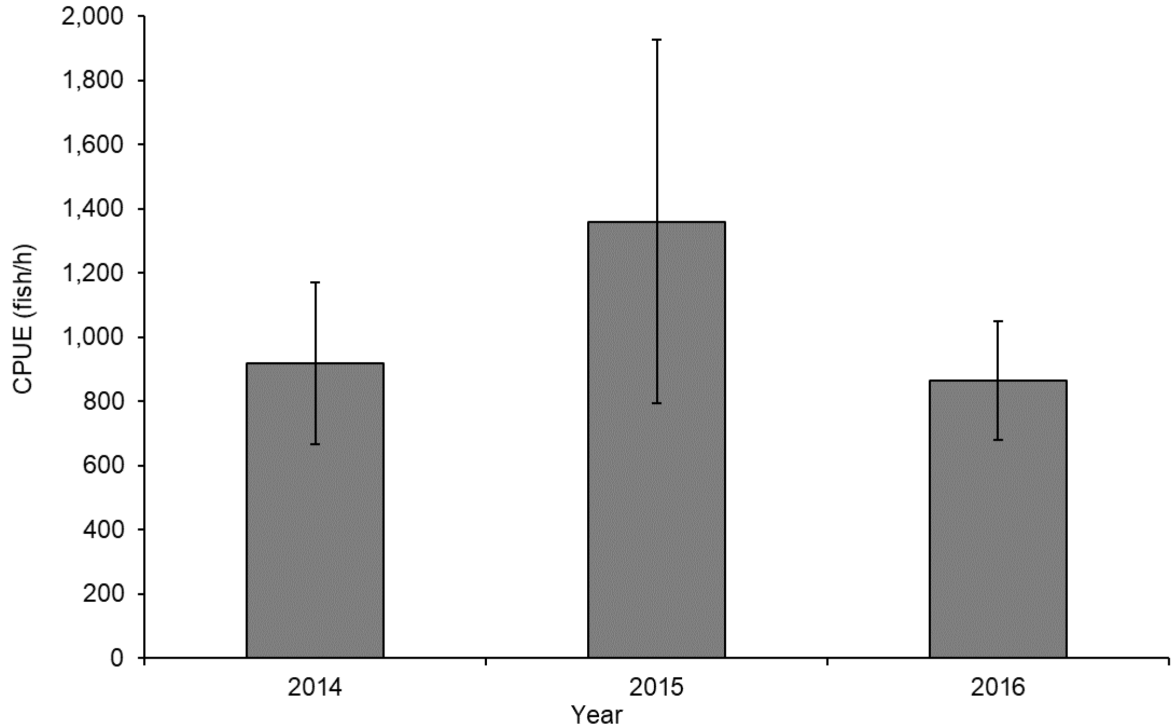


Figure 15. Catch-per-unit-effort (CPUE; fish/h) of fishes sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2016. Error bars represent 90% confidence intervals.

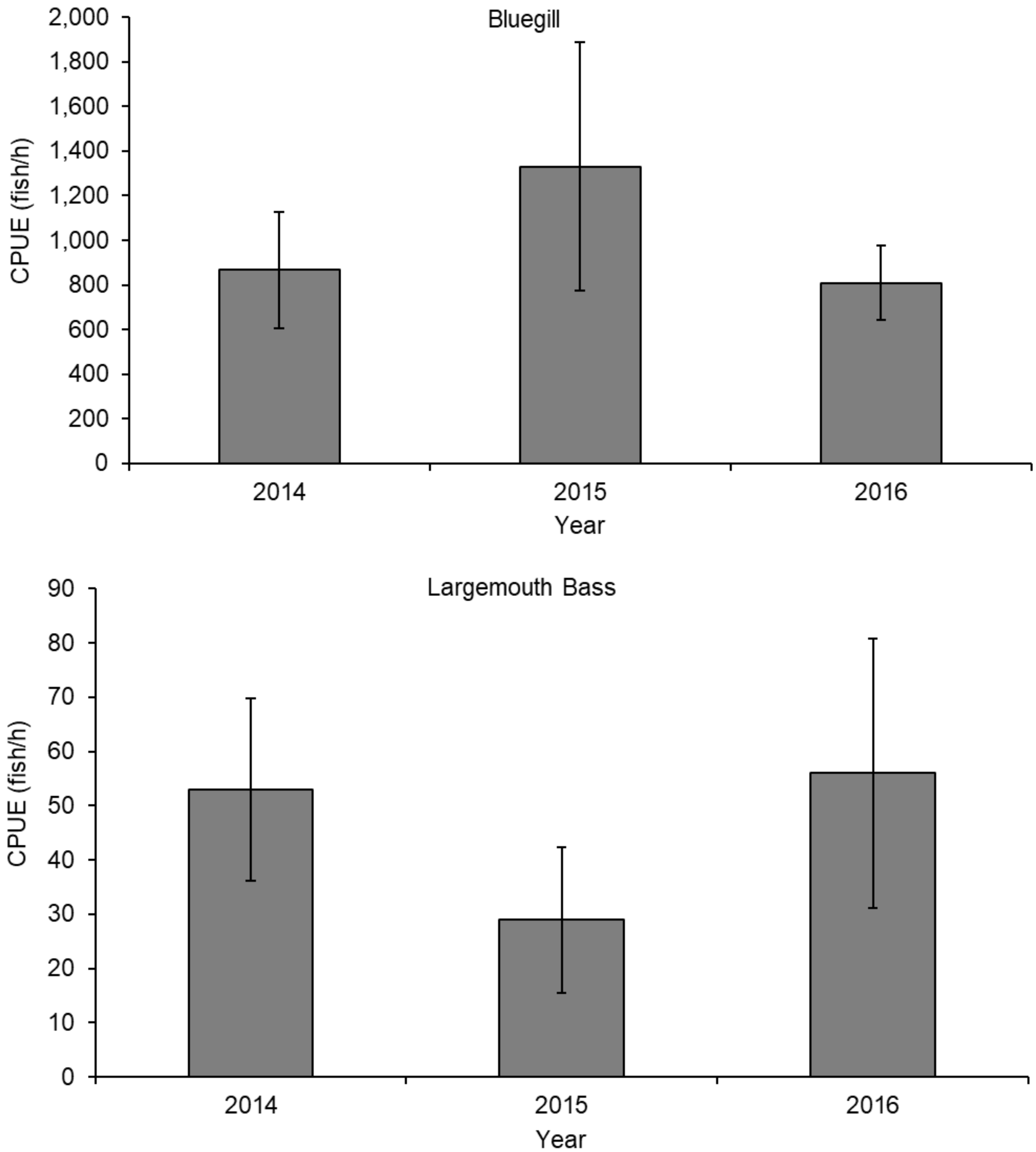


Figure 16. Catch-per-unit-effort (CPUE; fish/h) by species, of fishes sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2016. Error bars represent 90% confidence intervals.

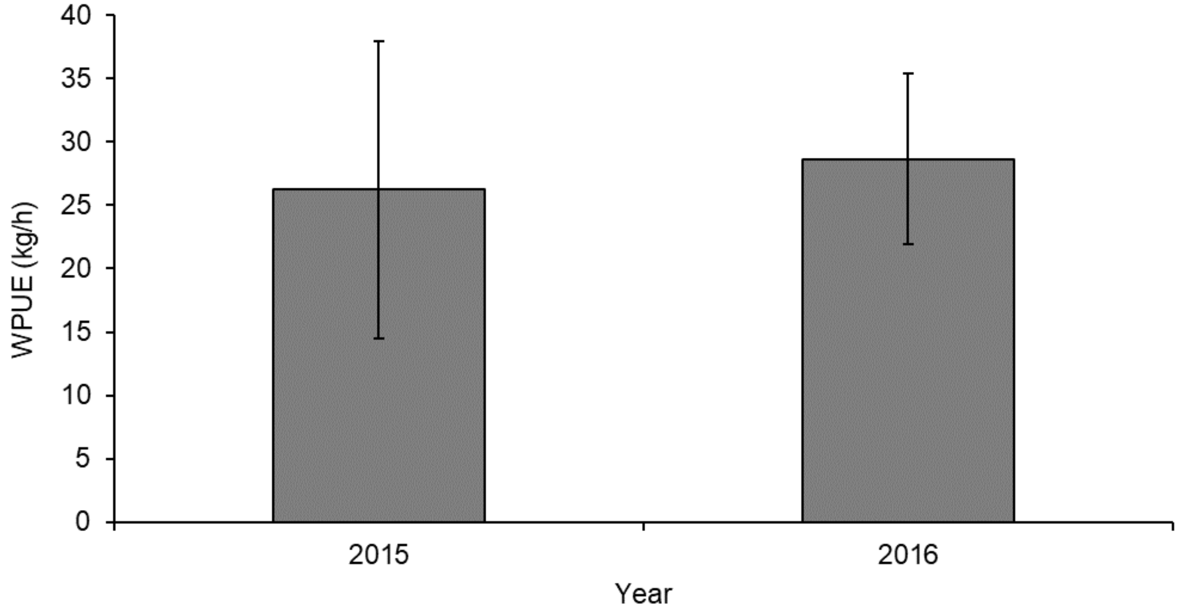


Figure 17. Weight-per-unit-effort (WPUE; kg/h) of fishes sampled by electrofishing Deyo Reservoir, Idaho, in 2015 and 2016. Error bars represent 90% confidence intervals.

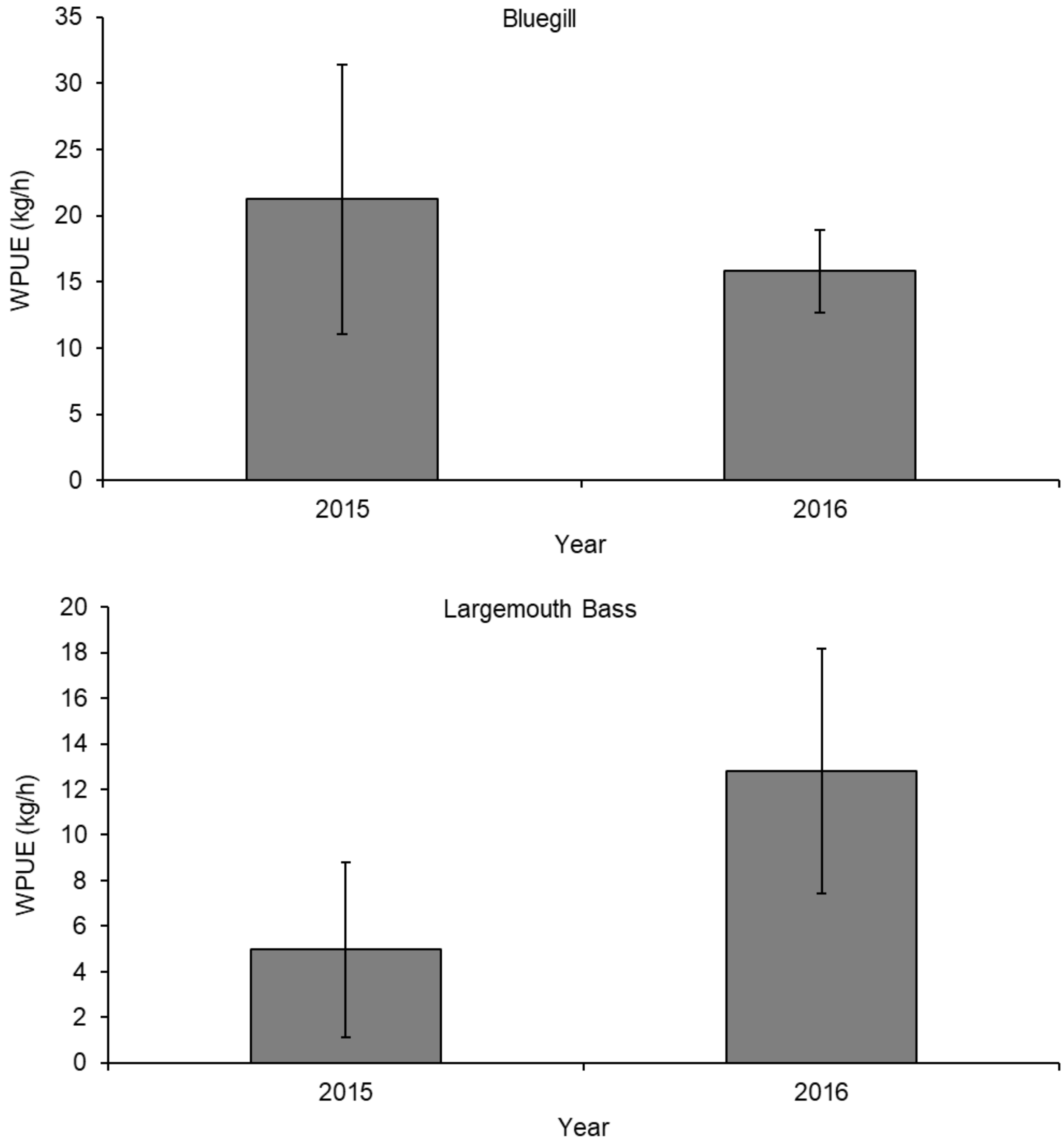


Figure 18. Weight-per-unit-effort (WPUE; kg/h) by species, of fishes sampled by electrofishing Deyo Reservoir, Idaho, in 2015 and 2016. Error bars represent 90% confidence intervals.

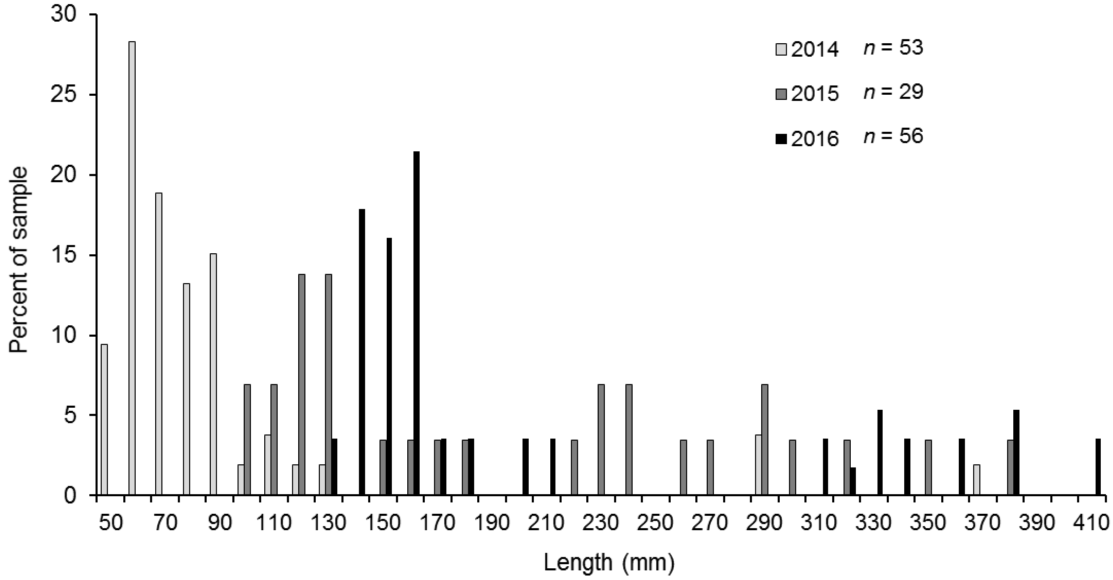


Figure 19. Length-frequency distribution of Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2016.

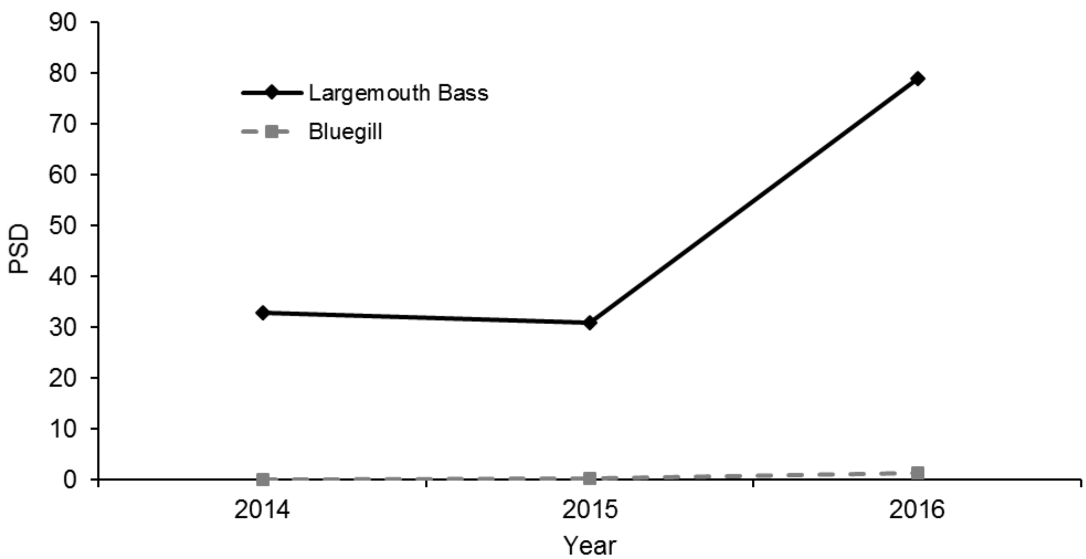


Figure 20. Proportional Size Distribution (PSD) values of Largemouth Bass and Bluegill sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2016.

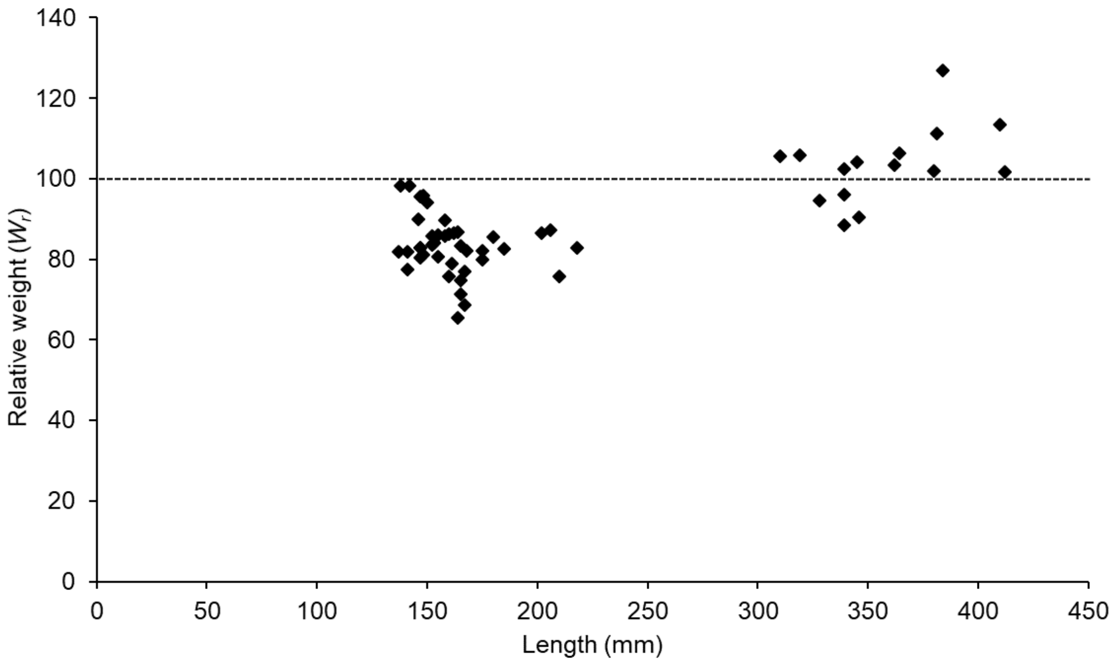


Figure 21. Relative weight (Wr) of Largemouth Bass sampled by electrofishing Deyo Reservoir, Idaho, in 2016.

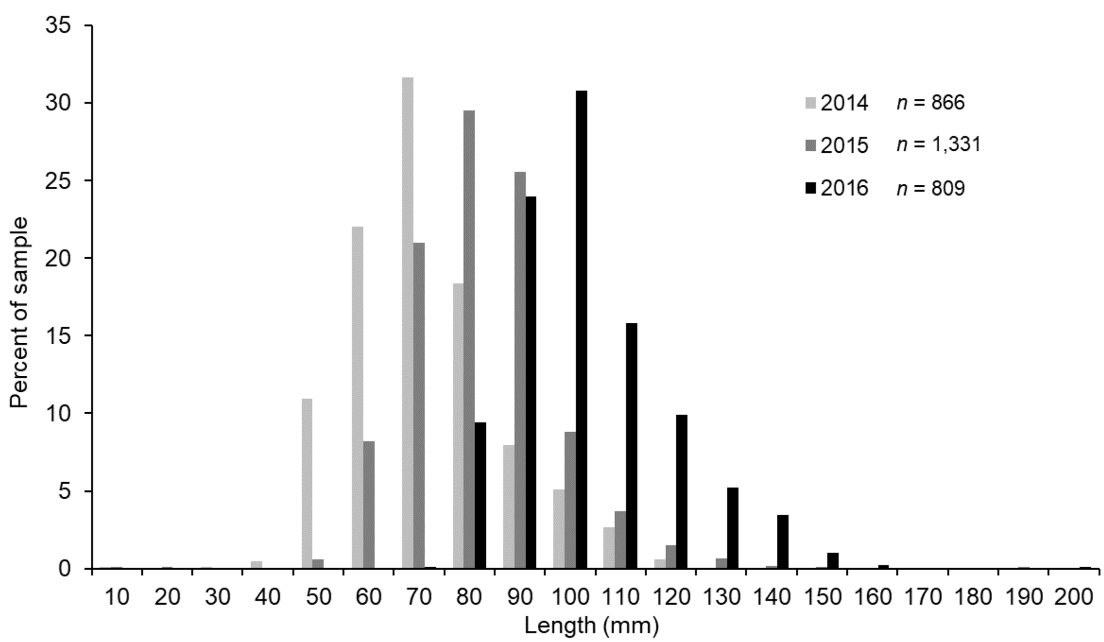


Figure 22. Length-frequency distribution of Bluegill sampled by electrofishing Deyo Reservoir, Idaho, from 2014 to 2016.

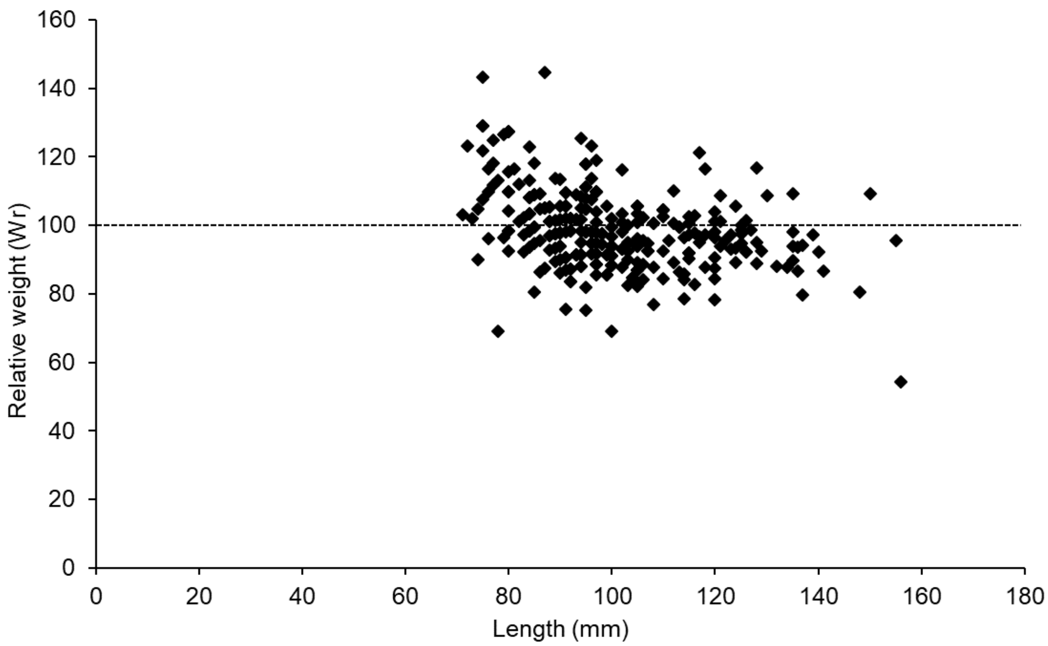


Figure 23. Relative weights (Wr) of Bluegill sampled by electrofishing Deyo Reservoir, Idaho, in 2016.

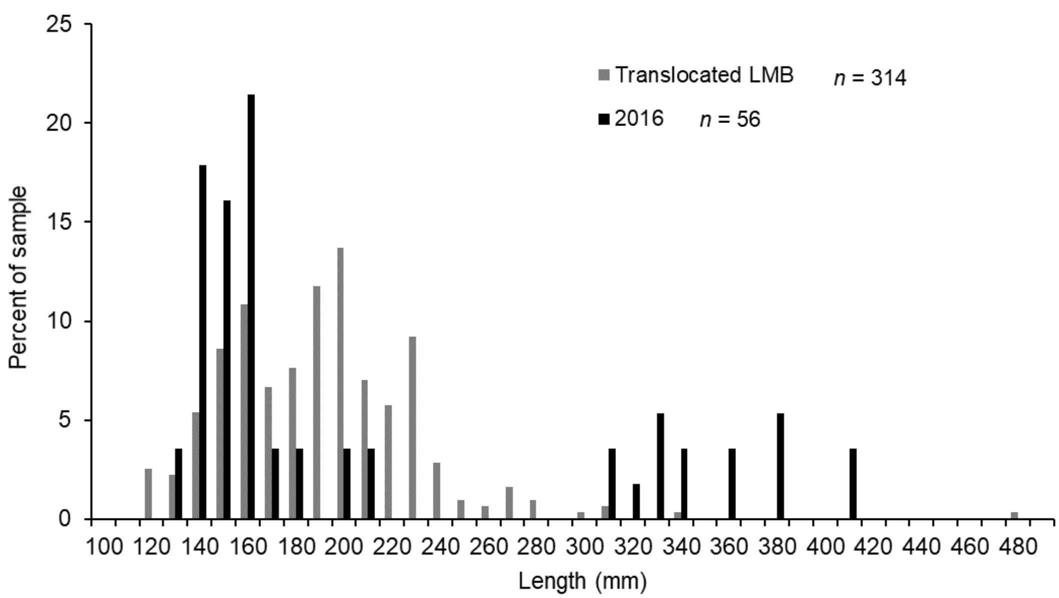


Figure 24. Comparison of length-frequency distributions of Largemouth Bass (LMB) sampled by electrofishing Deyo Reservoir, Idaho, on May 17, 2016, to LMB collected from Smith Lake and Bonner Lake, Idaho, and translocated into Deyo Reservoir on June 17, 2016.

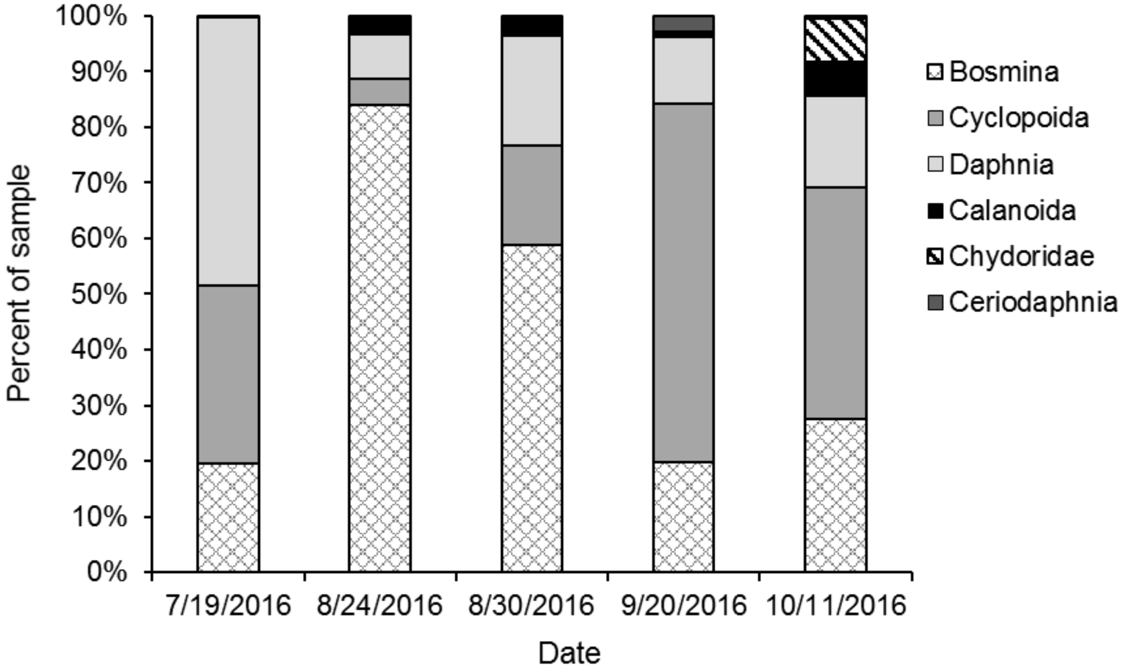


Figure 25. A comparison of zooplankton community composition in Deyo Reservoir, Idaho, based on monthly samples collected in 2016.

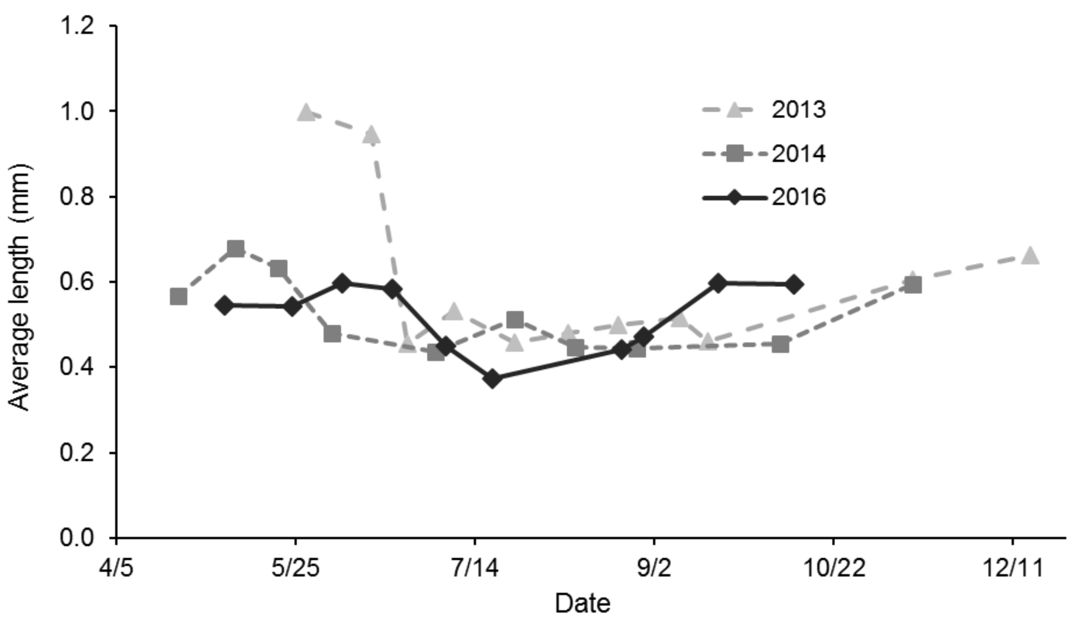


Figure 26. Average length of Daphnia in zooplankton samples collected during 2013, 2014, and 2016 in Deyo Reservoir, Idaho.

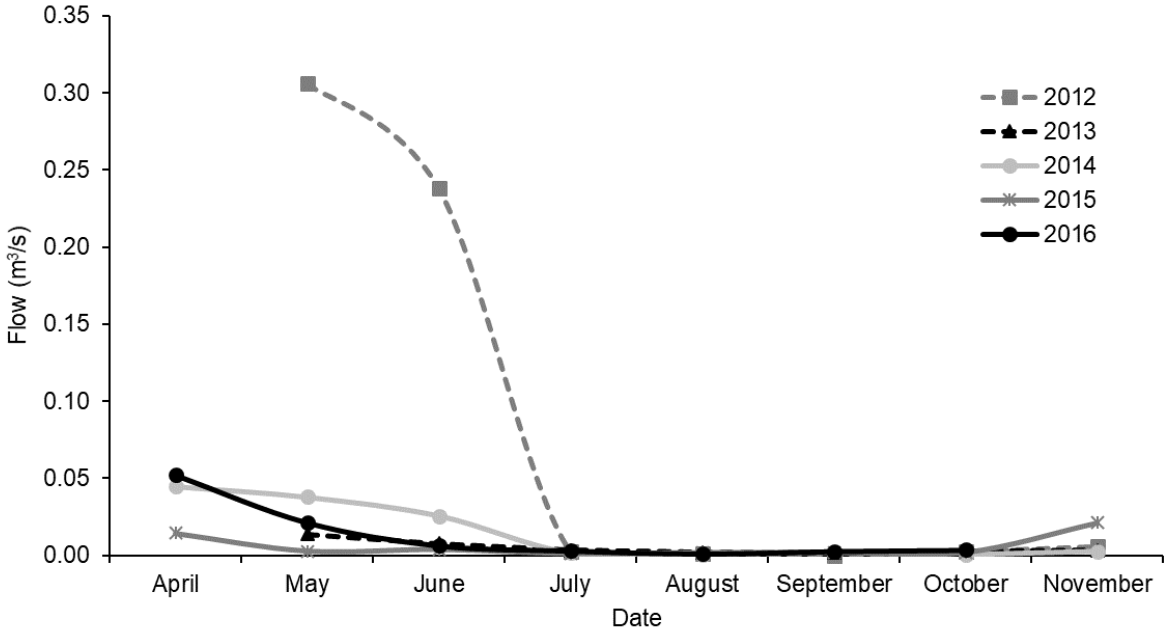


Figure 27. Average monthly flow ($m^3/s$; cubic meters per second) at the Lower Station on Schmidt Creek, Idaho, from 2012 to 2016.

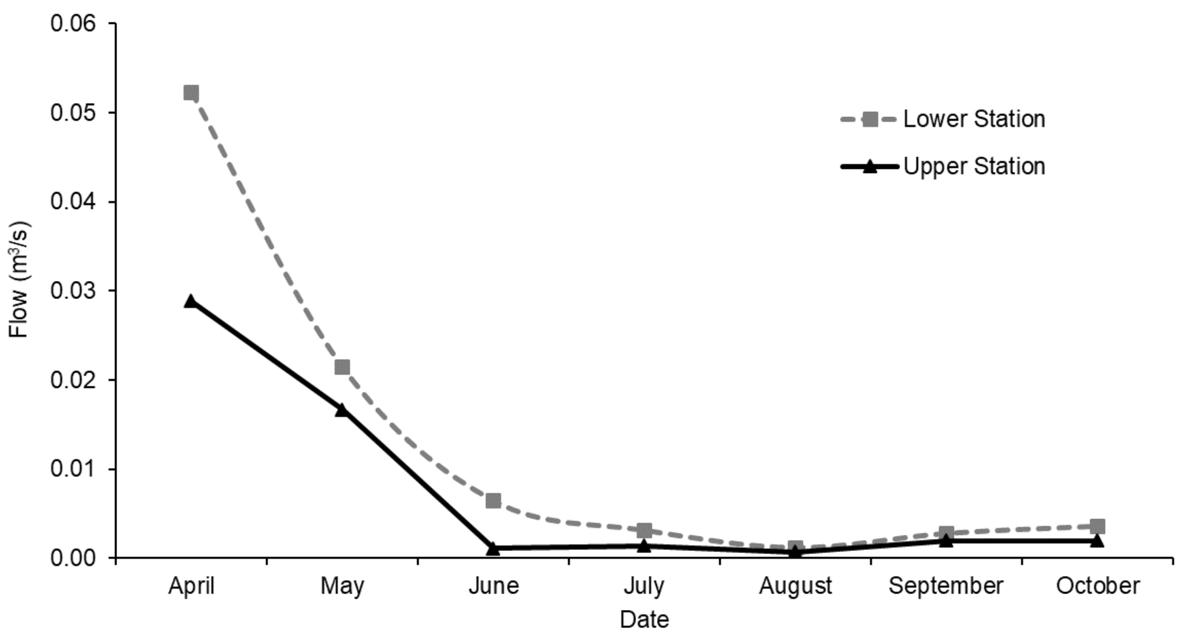


Figure 28. Average monthly flow (m3/s; cubic meters per second) at two sampling stations on Schmidt Creek, Idaho, in 2016.

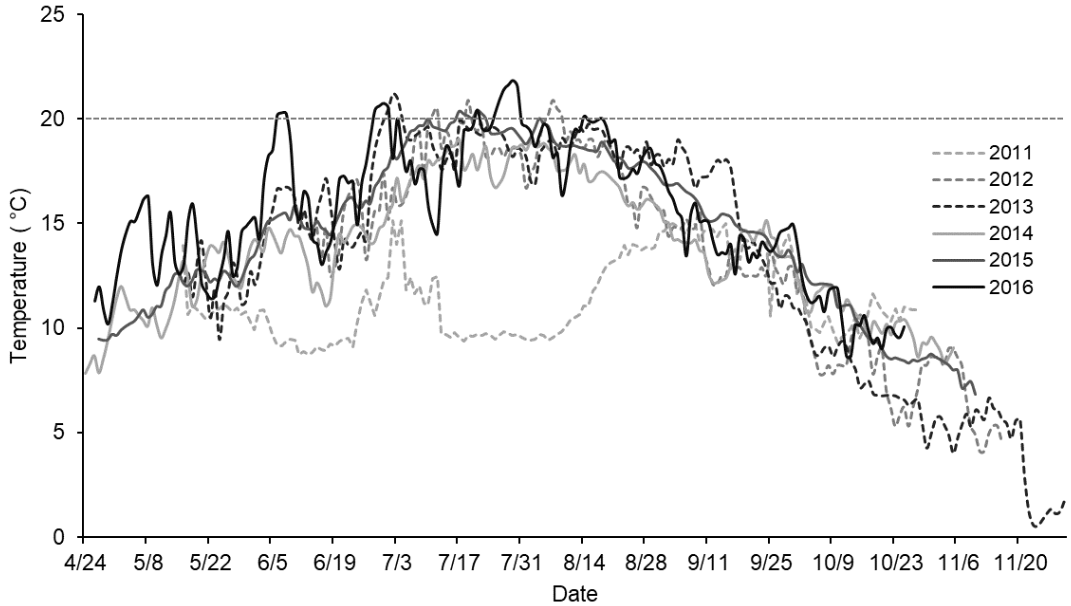


Figure 29. Daily maximum water temperatures measured at Lower Station on Schmidt Creek, Idaho, monitoring station from 2011 to 2016. The dashed horizontal line represents 20°C, a temperature that steelhead have been found to avoid.

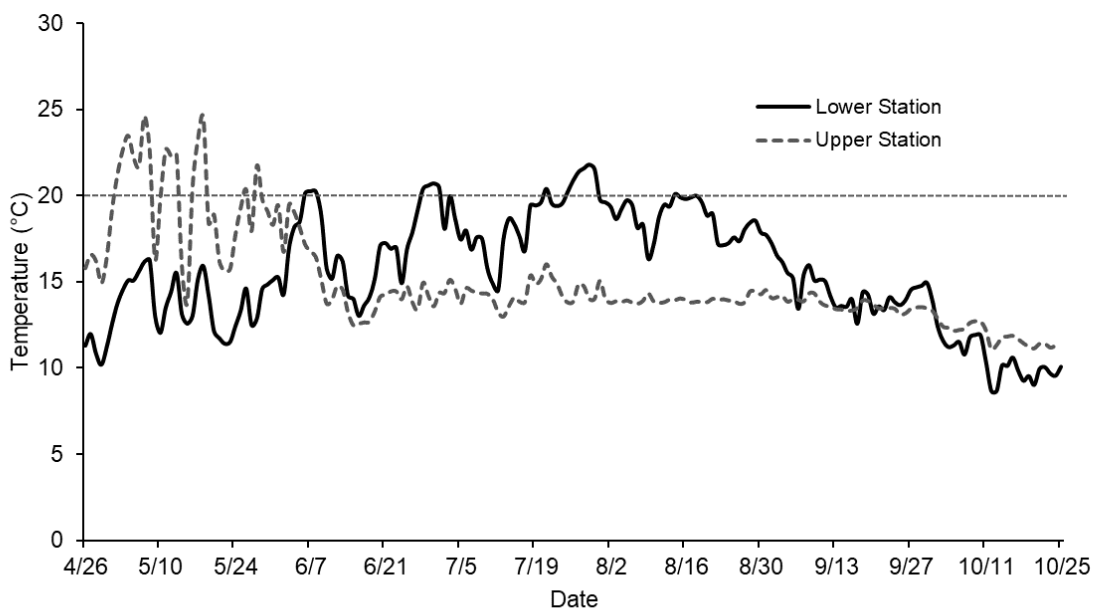


Figure 30. Comparison of maximum daily water temperature in Schmidt Creek, Idaho, between the Lower Station and Upper Station from April 26th to October 25th, 2016. The dashed horizontal line represents 20°C, a temperature that steelhead have been found to avoid.

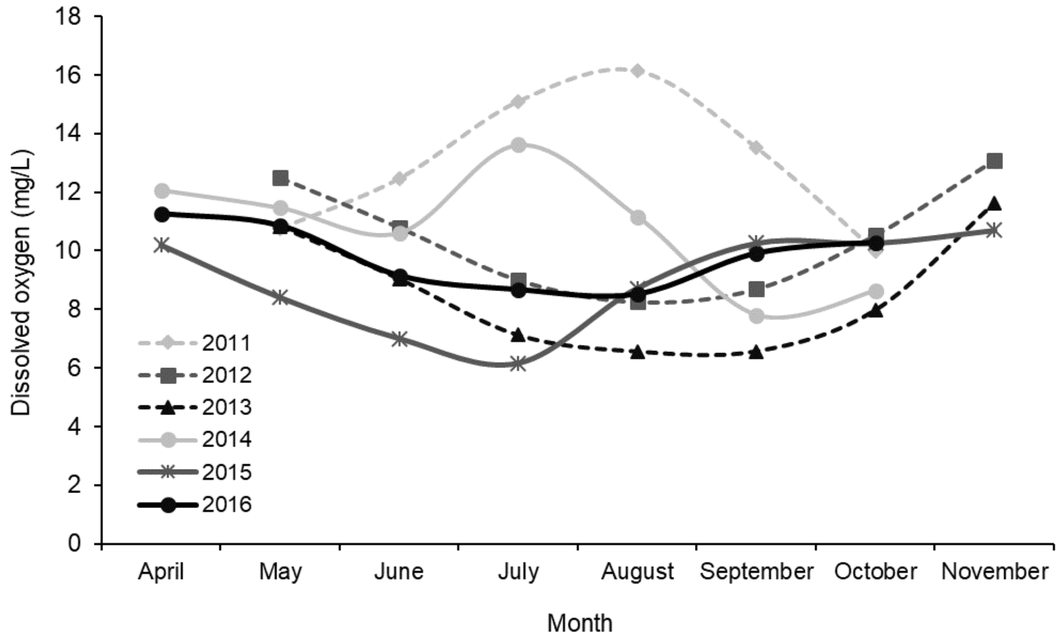


Figure 31. Average monthly dissolved oxygen levels at the Lower Station on Schmidt Creek, Idaho, monitoring station during April - November, from 2011 to 2016.

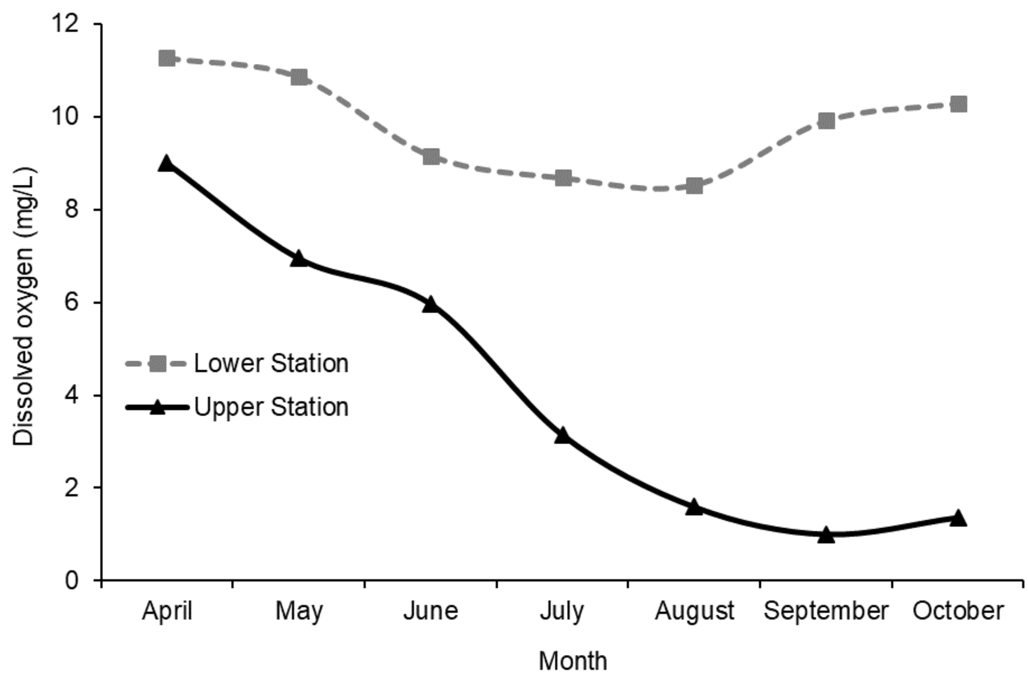


Figure 32. Comparison of average monthly dissolved oxygen levels between the Lower Station and Upper Station in Schmidt Creek, Idaho, during 2016.

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## DWORSHAK RESERVOIR CREEL SURVEYS

### ABSTRACT

A creel survey was conducted on Dworshak Reservoir from April 1 to August 31, 2016. Anglers fished an estimated 28,323 days and 156,553 h, caught 267,647 fish, and harvested 150,341, including 133,188 kokanee, 12,299 Smallmouth Bass, and 2,796 other species. The catch rate for kokanee anglers (1.6 fish/h) was the highest on record and the size of harvested kokanee (mean = 261 mm TL) was slightly above average. The catch rate for bass (2.1 fish/h) and the size of harvested bass (mean = 348 mm TL) were also the highest on record. Indications are that the current management of these fisheries is sustainable, but the bass fishery should be monitored to ensure that the trophy component is maintained.

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## INTRODUCTION

Dworshak Reservoir was the most popular fishing destination in Clearwater County and the second most popular destination in the Clearwater region, based on total angler trips in 2011 (Thomas MacArthur, IDFG, unpublished data). It provides a multi-species fishery for naturally-reproducing kokanee *Oncorhynchus nerka*, Smallmouth Bass *Micropterus dolomieu*, and Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi*, as well as hatchery-stocked Rainbow Trout *Oncorhynchus mykiss*. The reservoir also provides habitat for Bull Trout *Salvelinus confluentus*, which are listed as Threatened under the Endangered Species Act (ESA). A creel survey was conducted in 2016 to assess the status and trends of this fishery.

## OBJECTIVES

1. Estimate angling effort, harvest, and catch rates for the fishery as a whole and for the two predominant target species, kokanee and Smallmouth Bass.

2. Assess trends in these performance measures in relation to current management.

## STUDY AREA

Dworshak Reservoir was impounded after the construction of Dworshak Dam in 1972 on the North Fork Clearwater River approximately 2.4 km from its confluence with the mainstem Clearwater River. The reservoir is narrow, steeply sloped, and primarily surrounded by coniferous forests. The North Fork Clearwater River and its tributaries drain nearly 632,000 ha, which is composed primarily of montane forests in steeply sloped terrain (Falter et al. 1977). The underlying geology is composed of Columbia River basalt and metamorphic sediments with granitic intrusions covered by shallow soils (Falter et al. 1977). Most of the North Fork Clearwater watershed above the reservoir lies within the Clearwater National Forest. The reservoir is immediately surrounded by land managed by the US Army Corps of Engineers (USACE), but much of the lower watershed is privately owned. Timber harvest is the primary commercial activity, although there is some agriculture in the lower watershed.

At full pool, Dworshak Reservoir is 86.3-km long with a surface area of 6,916 ha and a volume of 4.3 billion $m^3$ (Falter 1982). Typical annual drawdown lowers the pool elevation by 24 m and reduces the surface area by 27%. Peak pool elevation is typically reached by late June and drawdown begins after the first week of July, with typical minimum pool elevation reached by the second week of September. The mean hydraulic retention time is 10.2 months (Falter 1982) and the mean daily discharge from 2004-2015 was 151 $m^3/s$ (http://www.cbr.washington.edu/dart/; accessed 1/12/16). Historically, Dworshak Reservoir begins to thermally stratify in April and stratification becomes pronounced from June through September. Destratification begins in the fall and occurs more rapidly at the upper end of the reservoir (Falter 1982, Wilson and Corsi 2016).

A nutrient restoration program has been conducted since 2007 in cooperation with the U.S. Army Corps of Engineers. This program has increased both the productivity and efficiency of the foodweb by adding nitrogen (N) in the form of ammonium nitrate on a weekly basis from May through September. The addition of N has promoted the growth of edible phytoplankton instead of inedible and potentially toxin cyanobacteria, increased the biomass of Daphnia, an important prey source for planktivorous fish, and increased the growth of kokanee at a given density. Increasing the productivity of the kokanee population is expected to improve the performance of both the kokanee fishery, as well as fisheries for piscivorous fish that feed on kokanee (i.e. Smallmouth Bass).

With the exception of Cutthroat Trout and Bull Trout, there are no special fishing rules for Dworshak Reservoir. The daily bag limit for kokanee is 25, with no minimum length. The daily limit for bass is six, with no minimum length. The daily limit for trout is six, with no minimum length. However, only two trout may be Cutthroat Trout, none of which may be under 14 inches, 25 Brook Trout may be retained, and no harvest of Bull Trout With the exception of Smallmouth Bass, these rules have received little if any scrutiny from the angling public. However, a number of bass anglers have expressed concerns that current regulations may result in a loss of opportunity for trophy Smallmouth Bass fishing.

## METHODS

Creel surveys were conducted consistently from April 1 to August 31, 2016. For these surveys, we used an access-access design (Pollock et al. 1994). The survey was stratified by month and day type (weekday or weekend/holiday). Sampling days, locations, and shifts were chosen at random. Days within a strata were given equal selection probabilities and two days were randomly chosen from each strata per week. The available daylight was divided into two shifts (am or pm) of equal length and shift probability was based on the relative number of interviews obtained during each time period from previous years. Access to the reservoir, whether by boat or shore, was limited to six locations (Figure 33). The two boat ramps at Bruce's Eddy were treated as separate access points and given independent selections probabilities, as were the ramp and marina at Big Eddy. Access points were assigned selection probabilities based on the relative number of interviews obtained during each in previous years and whether or not a ramp was usable at the time (ramp availability changed with pool elevation).

Creel clerks were instructed to remain at an assigned access point during the entire length of the shift, or until all boat trailers and shore anglers were gone in the case of a pm shift. They were further instructed to make every effort to interview every party returning to the access site by boat, or departing from the access site by vehicle in the case of shore anglers. In the event that an interview could not be obtained, clerks recorded the party as unknown and noted the time of return. Lengths were collected from a random subsample of harvested fish.

In an access-access survey design, data are only collected for completed trips and total effort is estimated by expanding the effort documented for anglers returning to a given access point during a given shift by the probability of selecting that location and shift (Pollock et al. 1994). Daily effort ($\hat{e}_d$), measured in angler hours, was estimated in the following manner:

$$\hat{e}_d = \frac{e_{rsd}}{(\pi_r \times \pi_s \times \pi_b)}$$

Where: $\hat{e}_d$ = Estimated total fishing effort for day *d*.
$e_{rsd}$ = Fishing effort sampled at site *r*, during shift *s*, on day *d*.
$\pi_r$ = Selection probability of access site *r*.
$\pi_s$ = Selection probability of shift *s*.
$\pi_b$ = Probability of sampling a given boat during that shift.

The probability of sampling a given boat during a particular shift was simply calculated as the ratio of the number of boats sampled during that shift (including those that were not fishing) over the number returning (including those that were not sampled). Effort was also estimated in terms of angler days, which was calculated in a like manner. Angler days were calculated by summing the number of anglers fishing on a given day, irrespective of the time spent during the course of that day.

Total effort for a given strata was calculated by multiplying the mean daily effort for that strata by the number of days in the strata. Monthly effort was calculated by summing the effort of the strata within each month, and annual effort was calculated by summing the monthly effort.

Total catch and total harvest were estimated in the same manner as effort, substituting each into the above formulas. Formulas used to calculate standard errors for catch and effort can be found on pages 234-236 of Pollock et al. (1994). Catch rates were calculated by dividing total catch for the respective period by total effort. In addition, we calculated these metrics for anglers that just targeted kokanee or Smallmouth Bass.

Comparisons of effort, harvest, catch rate, and mean size of harvested fish were also compared to previous years. However, since the length of the survey has not been consistent between years, we only compared a timeframe (April through July) that has been sampled consistently. This time frame encompasses nearly all of the effort directed toward kokanee, but misses effort directed toward bass during the late summer and fall.

## RESULTS

There were 1,107 interviews collected over 86 creel shifts for Dworshak Reservoir between April 1 and August 31, 2016. From these, we estimated anglers fished 28,323 days (SE = 4,547 or 16%) or 156,553 h (SE = 30,096 or 19%; Table 3). Fishing effort peaked in July with 6,354 angler days (SE = 1,426 or 22%) and 45,723 angler h (SE = 14,928 or 33%; Table 3). We documented the least effort in April with 1,696 angler days (SE = 237 or 14%) and 19,781 angler h (SE = 3,101 or 16%). The mean length of a single day of fishing was 4.4 h with a range of 0.5-15 h. Fishing party sizes ranged from 1 to 11 anglers with a mean party size of 3.4. Anglers caught 267,647 fish (SE = 78,368 or 29%), including 133,828 kokanee, 130,231 Smallmouth Bass, and 3,588 other species. Of the fish caught, 150,341 (SE = 41,742 or 28%) were harvested, including 133,188 kokanee (100% of catch), 12,299 Smallmouth Bass (9% of catch), and 2,796 other species (78% of catch).

Anglers specifically targeting kokanee fished 15,381 days (SE = 3,856 or 25%) or 81,216 h (SE = 21,806 or 27%). Kokanee fishing effort was relatively low in April and May, increased in June and July, and then dropped off in August (Table 4). Effort directed toward kokanee from April through July of 2016 was higher than the two previous years (Figure 35). Kokanee anglers caught an estimated 132,599 kokanee (SE = 40,119 or 30%) and harvested 131,894 (SE = 40,000 or 30%), with a mean catch rate of 1.6 fish/h. Catch rates for kokanee were lowest in April (mean = 0.3 fish/h), increased to a high in June (mean = 2.1 fish/h), then declined through August (mean = 1.3 fish/h). Both harvest and catch rate were greater for 2016 than the two previous years (Figure 36). Most kokanee anglers (92%) caught at least one kokanee during a given day. Of those who harvested at least one kokanee, 6% harvested a limit of 25 kokanee. The mean length of harvested kokanee was 261 mm TL (Figure 37), which was greater than the two previous years (Figure 38). Kokanee anglers also caught an estimated 7,464 incidental species (SE = 5,898), all of which were Rainbow Trout or Smallmouth Bass, and harvested 2,441 of these (SE = 1,605).

Anglers specifically targeting Smallmouth Bass fished 10,740 days (SE = 3,398 or 32%) or 57,403 h (SE = 20,783 or 36%). Bass fishing effort was similar in most months we surveyed except for a dip in June (Table 4). Effort directed toward bass from April through July of 2016 was greater than 2014, but less than 2015 (Figure 34). Bass anglers caught an estimated 119,491 Smallmouth Bass (SE = 54,369 or 46%) for a mean catch rate of 2.1 fish/h, and harvested 10,421 (SE = 4,924 or 47%). Catch rates for Smallmouth Bass were lowest in April (mean = 0.7 fish/h), increased to a high in August (mean = 4.4 fish/h). Harvest was greater for 2016 than the two previous years (Figure 38). Most bass anglers (86%) caught at least one bass during a given day. Of these, 31% harvested at least one fish, and 10% of these anglers harvested a limit of six bass. The mean length of harvested bass was 348 mm TL for the entire survey (Figure 36), and 354 mm TL from April through July, which was greater than the two previous years (Figure 37). The mean size of harvested bass was greatest in April (mean = 371 mm TL) and steady decreased through August (mean = 291 mm TL). Bass anglers also caught an estimated 2,392 incidental species (SE = 1,399 or 58%), and harvested 2,008 of these (SE = 1,374 or 68%).

## DISCUSSION

Kokanee fishing on Dworshak Reservoir was exceptional in 2016 compared to most other years. The catch rate for kokanee anglers was the highest that has been documented (Wilson and Corsi 2016) and harvest increased considerably over the previous two years. The mean length of harvested kokanee also increased over the previous two years and was slightly higher than average. The performance of the 2016 kokanee fishery on Dworshak Reservoir was in large part due to a record year class of age-3 fish (Wilson and Corsi 2018). Due to the extreme abundance of this year class (in 2015), these fish were small (202 mm TL) at age-2, the age at which kokanee typically spawn in this system. This is likely to have resulted in a higher proportion of this year class maturing at age-3 (Grover 2005), resulting in more abundant and larger kokanee than average in 2016.

The fishing seasons and rules for kokanee have had little influence on the long-term viability of the kokanee fishery on Dworshak Reservoir. The current regulations restrict kokanee harvest very little as just 6% of anglers harvest a limit, resulting in annual harvest of around 25% of the spawning-sized fish. However, strong year classes of kokanee, such as the one that produced the record number of age-2 and age-3 kokanee in 2015 and 2016, came from the lowest numbers of spawners documented since 2000 (Wilson and Corsi 2016). Therefore, this fishery is not likely to be recruitment-limited. Likewise, Askey and Johnston (2013) found that the kokanee fishery in Okanagan Lake was self-regulating and that variation in abundance was driven by variation in stock productivity, not harvest. The nutrient restoration program that began in 2007 has created growing conditions in the reservoir that make it possible to support higher abundances of kokanee than it once could (Wilson and Corsi 2016). Although entrainment is still likely to cause occasional collapses in the kokanee population, such as in 1997 and 2010 (Bennett 1997, Wilson et al. 2018).

The Smallmouth Bass fishery was also quite good compared to other years. The catch rate for bass anglers was higher in 2016 than the previous two years. While harvest was down from 2015, it was still the second highest on record. Moreover, the average size of harvested smallmouth was the highest on record. While the number and size of bass harvested has increased during the last two years, exploitation has remained relatively low (Smallmouth Bass Investigations, this report). Stable exploitation in conjunction with increased harvest is evidence that the Smallmouth Bass population is healthy and growing in Dworshak Reservoir.

Our analysis suggests that current regulations are sufficient to ensure the long-term viability of the Smallmouth Bass fishery on Dworshak Reservoir. The primary evidence for this is relatively stable harvest and increasing fish size (this section), along with stable exploitation (see Smallmouth Bass Investigations, this report). Most bass anglers practice catch-and-release fishing on Dworshak Reservoir. Volunteer catch and release has been reported as an increasing trend in black bass fisheries across North America since approximately 1975 (Noble 2002) and is common in other Idaho Smallmouth fisheries (McClure 2018). Of those who harvested bass on Dworshak Reservoir, only a small proportion harvested a limit of bass. Therefore, current bag limits do not appear to limit harvest. However, this is an important trophy fishery due to the large size attained by bass in the reservoir, including two state records. In order to maintain the trophy component of this fishery, both the fishery and bass population should be monitored so that regulations can be adjusted accordingly if warranted in the future.

## MANAGEMENT RECOMMENDATIONS

1. Maintain current fishing regulations for Dworshak Reservoir.

2. Continue future monitoring of the Smallmouth Bass population and fishery.

Table 3. Angler effort (reported as angler days and hours), catch and harvest estimated from a creel survey of Dworshak Reservoir from April 1 to August 31, 2016. Estimates are reported for all anglers combined.

| | Angler | | Kokanee | | Bass | | Other | |
|---|---|---|---|---|---|---|---|---|
| | Days | Hours | Caught | Kept | Caught | Kept | Caught | Kept |
| April | 3,603 | 19,781 | 2,141 | 2,074 | 8,748 | 1,772 | 757 | 418 |
| May | 3,670 | 20,389 | 5,277 | 5,233 | 21,536 | 1,457 | 360 | 274 |
| June | 8,054 | 41,810 | 65,679 | 65,743 | 15,068 | 1,529 | 333 | 176 |
| July | 9,015 | 45,723 | 54,850 | 54,850 | 19,467 | 1,381 | 517 | 350 |
| August | 3,981 | 28,850 | 5,880 | 5,288 | 65,412 | 6,161 | 1,621 | 1,577 |
| Totals | 28,323 | 156,553 | 133,827 | 133,188 | 130,231 | 12,300 | 3,588 | 2,795 |

Table 4. Angler effort (reported as angler days and hours), catch, catch rates, and harvest and mean total length (TL) of harvested fish estimated from a creel survey of Dworshak Reservoir from April 1 to August 31, 2016. Estimates are reported separately for anglers who only targeted kokanee or Smallmouth Bass (Bass).

| | Angler | | Target species | | | Incidental species | | |
|---|---|---|---|---|---|---|---|---|
| | Days | Hours | Caught | (Fish/h) | Kept | Caught | Kept | TL (mm) |
| | | | | **Kokanee anglers** | | | | |
| April | 1,504 | 7,507 | 2,124 | 0.3 | 2,056 | 153 | 418 | 228 |
| May | 1,067 | 4,592 | 4,973 | 1.1 | 4,968 | 951 | 274 | 243 |
| June | 5,847 | 30,893 | 64,772 | 2.1 | 64,731 | 354 | 176 | 262 |
| July | 6,166 | 33,562 | 54,850 | 1.6 | 54,850 | 500 | 350 | 268 |
| August | 798 | 4,662 | 5,880 | 1.3 | 5,288 | 5,520 | 1,577 | 263 |
| Total | 15,382 | 81,216 | 132,599 | 1.6 | 131,893 | 7,478 | 2,795 | 261 |
| | | | | **Bass anglers** | | | | |
| April | 2,062 | 12,018 | 8,576 | 0.7 | 1,686 | 461 | 275 | 371 |
| May | 2,172 | 13,884 | 18,816 | 1.4 | 1,156 | 241 | 177 | 369 |
| June | 1,742 | 7,015 | 13,585 | 1.9 | 1,450 | 251 | 161 | 350 |
| July | 2,616 | 10,795 | 18,649 | 1.7 | 1,337 | 80 | 80 | 306 |
| August | 2,147 | 13,691 | 59,865 | 4.4 | 4,791 | 1,358 | 1,314 | 291 |
| Total | 10,739 | 57,403 | 119,491 | 2.1 | 10,420 | 2,391 | 2,007 | 348 |

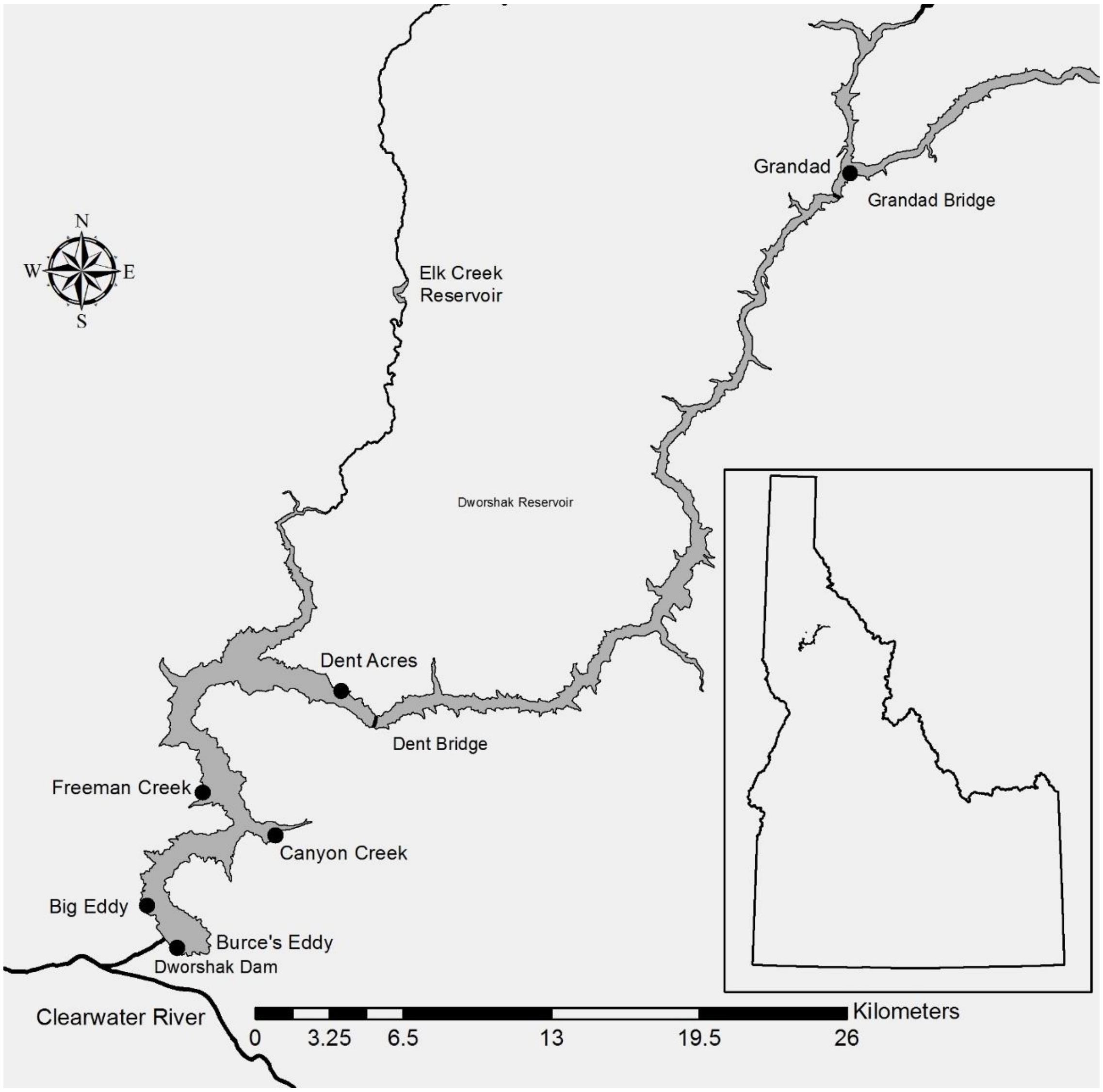



Figure 33. Fishing and boating access sites for Dworshak Reservoir, Idaho. Creel surveys were performed at all access sites in 2016.

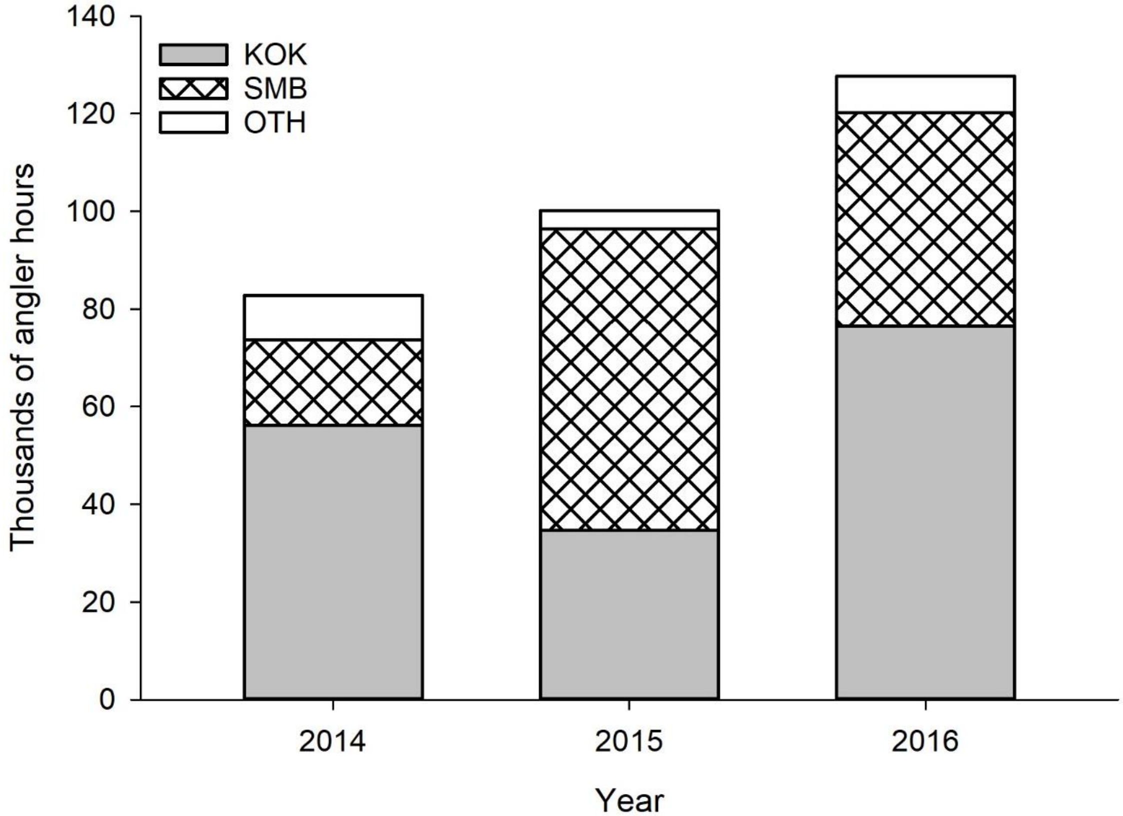



Figure 34. Fishing effort on Dworshak Reservoir from April through July of 2014 through 2016. Effort is divided by those targeting only kokanee, only Smallmouth Bass, or all other fishing effort.

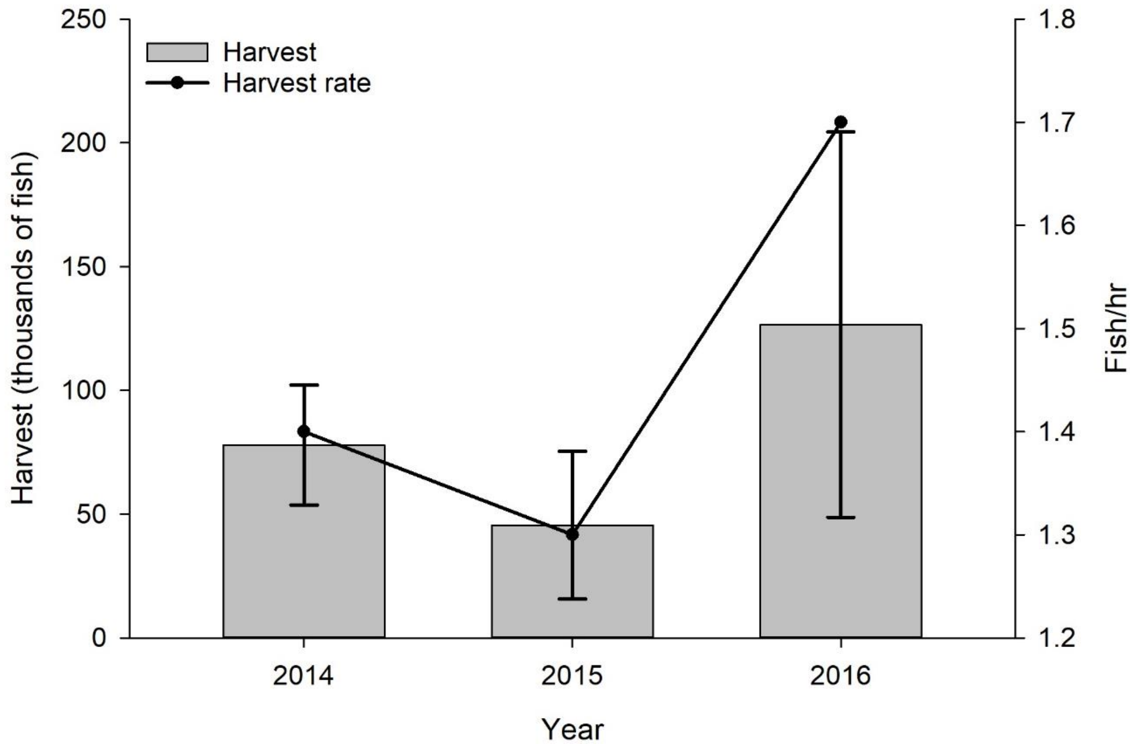
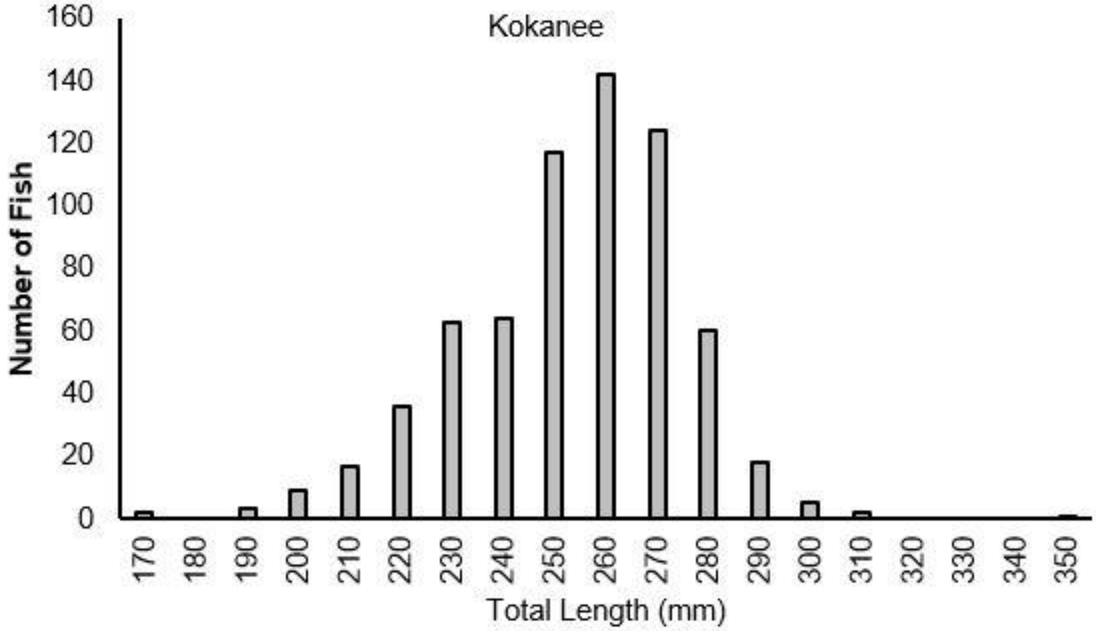



Figure 35. Harvest and harvest rates of kokanee for anglers targeting kokanee on Dworshak Reservoir from April through July of 2014 through 2016.



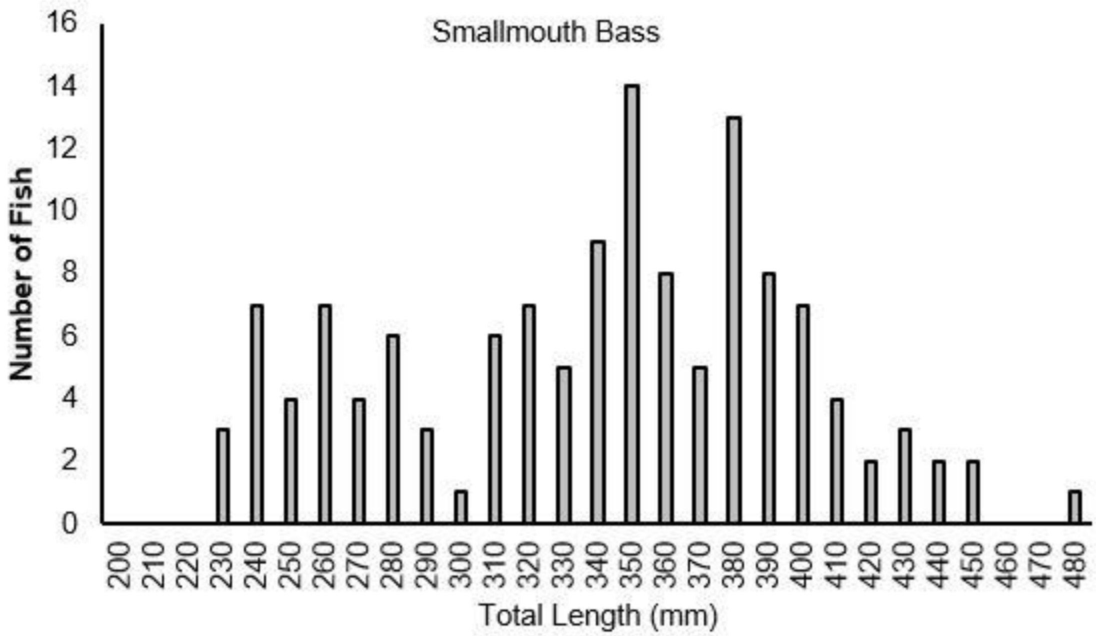




Figure 36. Length-frequency distributions for 663 kokanee and 131 Smallmouth Bass harvested from Dworshak Reservoir from April through August of 2016.

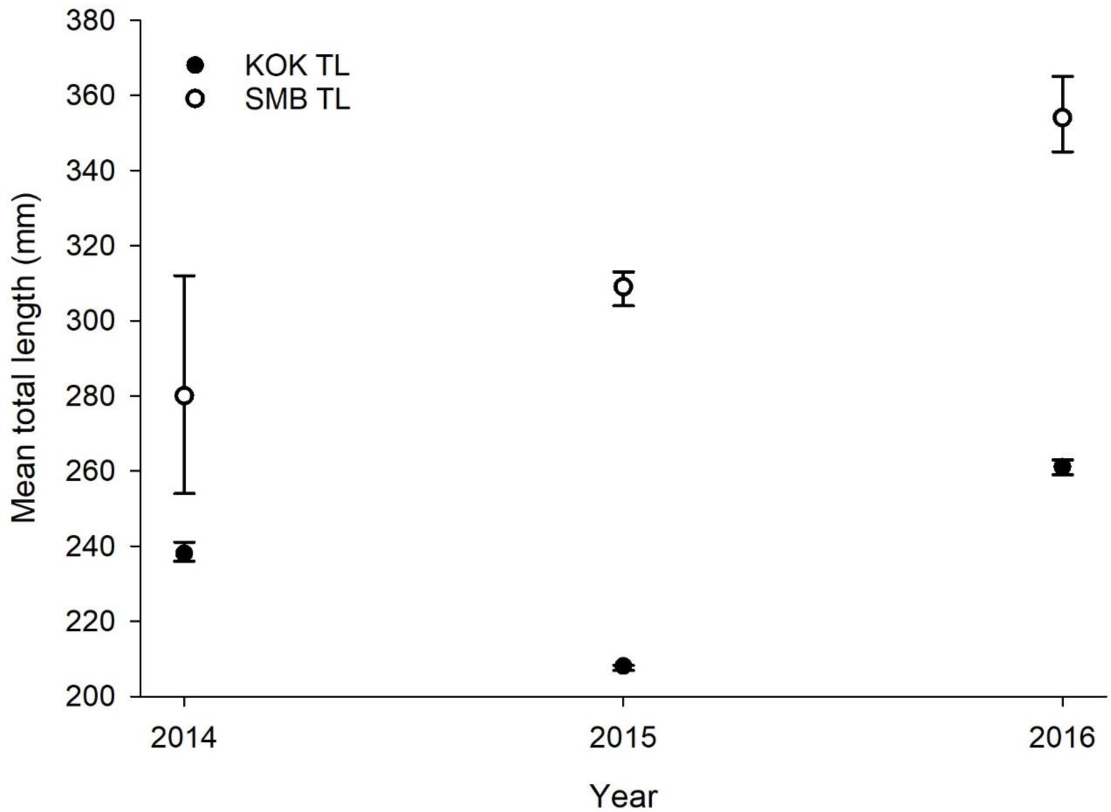



Figure 37. Mean length of kokanee and Smallmouth Bass harvested from Dworshak Reservoir from April through July of 2014 through 2016.

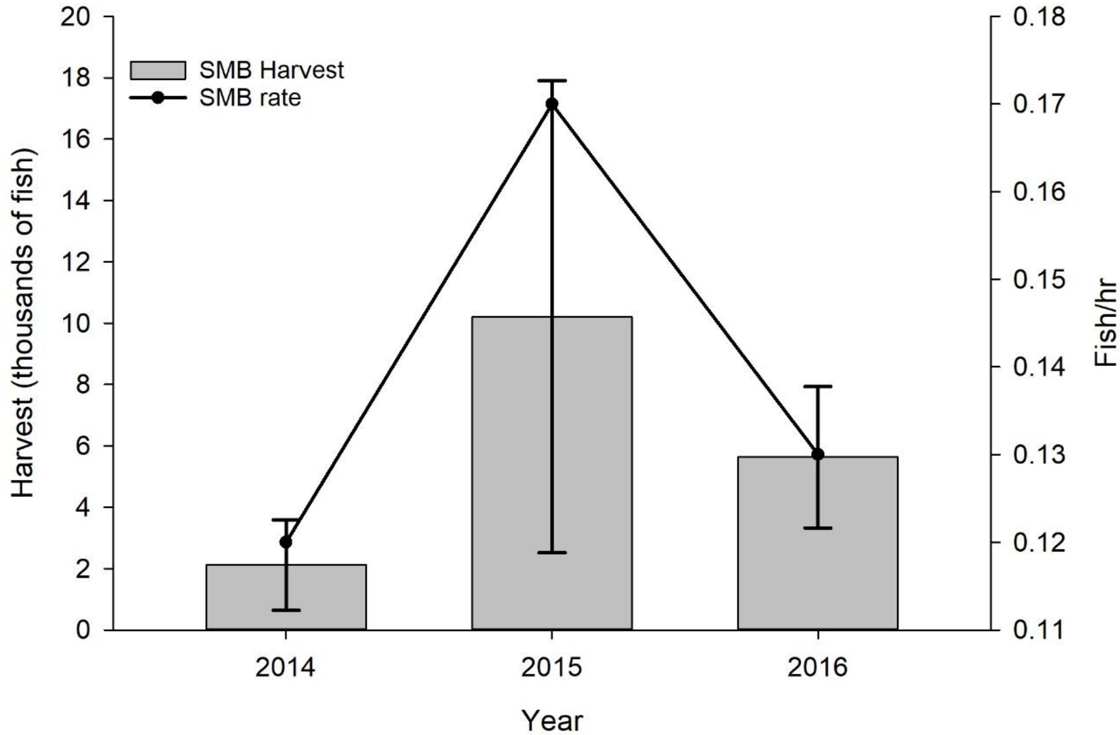



Figure 38. Harvest and harvest rates of Smallmouth Bass for anglers targeting bass on Dworshak Reservoir from April through July of 2014 through 2016.

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## DWORSHAK RESERVOIR SMALLMOUTH BASS INVESTIGATIONS

## ABSTRACT

Dworshak Reservoir provides a popular trophy fishery for Smallmouth Bass *Micropterus dolomieu*. Many bass anglers have been concerned that the quality of this fishery will decline due to overexploitation of larger fish. To assess the bass population and the effectiveness of current fishing rules, we conducted electrofishing surveys in 2015 and 2016 and tagged bass to estimate exploitation and abundance. To increase sample size, we also tagged bass collected by angling and at tournament weigh-ins from 2013 to 2016. The PSD of bass captured using electrofishing was highest when water temperatures were approximately 8°C and CPUE was highest when water temperatures were warmer. However, larger bass were still under represented with earlier timing of electrofishing surveys and estimates of mortality from these surveys are likely biased. The abundance of Smallmouth Bass in Dworshak Reservoir appears to be steady or increasing. While the PSD has declined in recent years, this appears to be due to increased recruitment of stock fish, rather than a decrease in the number of quality fish. The relative weight of bass declined as they grew from 200 to 300 mm TL, then increased until they reached 350 mm TL. On average, bass in Dworshak Reservoir grew to memorable length in 7 years and trophy length in 12 years, compared to the median for North America of 10 years to reach memorable length and more than 15 years to reach trophy length. Exploitation was highest for quality bass (21.3%) and lowest for memorable and trophy bass (7.1%). Due to rapid growth and low exploitation of larger bass, the current fishing rules are likely to sustain the trophy component of this fishery. Future work should be directed at obtaining reliable estimates of mortality and investigating factors that influence growth rates.

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## INTRODUCTION

Dworshak Reservoir provides an important resident fishery in Clearwater County and trophy-sized Smallmouth Bass *Micropterus dolomieu* are a popular component of this fishery. The last two state record Smallmouth Bass were caught in Dworshak Reservoir, and at least two other fish larger than the previous record are known to have been caught since the current record was established in 2006. The number of organized tournaments held on the reservoir increased from five to seven during the previous three years, to ten in 2016, including the finals for the Idaho Bass Federation. However, many bass anglers have expressed concerns of losing the trophy component of this fishery due to over exploitation, particularly of large fish, and have asked for more restrictive bass rules. Currently, there are no special rules for bass on Dworshak Reservoir and the general fishing regulations allow harvest of six bass/day of any size. The Smallmouth Bass population in Dworshak Reservoir was assessed to gain an understanding of its current state and assess whether current fishing rules are adequate to maintain the trophy component of this fishery.

## OBJECTIVES

1. Assess the effects of survey timing and water temperature on catch rates and length distributions obtained from electrofishing.

2. Estimate relevant population parameters for Smallmouth Bass in Dworshak Reservoir, including abundance, recruitment, growth, and exploitation.

## STUDY AREA

Dworshak Reservoir was impounded after the construction of Dworshak Dam in 1972 on the North Fork Clearwater River approximately 2.4 km from its confluence with the mainstem Clearwater River. The reservoir is narrow, steeply sloped, and primarily surrounded by coniferous forests. The North Fork Clearwater River and its tributaries drain nearly 632,000 ha, which is composed primarily of montane forests in steeply sloped terrain (Falter et al. 1977). The underlying geology is composed of Columbia River basalt and metamorphic sediments with granitic intrusions covered by shallow soils (Falter et al. 1977). Most of the North Fork Clearwater watershed above the reservoir lies within the Clearwater National Forest. The reservoir is immediately surrounded by land managed by the USACE, but much of the lower watershed is privately owned. Timber harvest is the primary commercial activity, although there is some agriculture in the lower watershed.

At full pool, Dworshak Reservoir is 86.3-km long with a surface area of 6,916 ha and a volume of 4.3 billion $m^3$ (Falter 1982). Typical annual drawdown lowers the pool elevation by 24 m and reduces the surface area by 27%. Peak pool elevation is typically reached by late June and drawdown begins after the first week of July, with typical minimum pool elevation reached by the second week of September. The mean hydraulic retention time was 10.2 months (Falter 1982) and the mean daily discharge was 154 $m^3/s$ from 2004 to 2014, 121 $m^3/s$ in 2015 and 145 $m^3/s$ in 2016 (http://www.cbr.washington.edu/dart/; accessed 5/25/18). Historically, Dworshak Reservoir begins to thermally stratify in April and stratification becomes pronounced from June through September. Destratification begins in the fall and occurs more rapidly at the upper end of the reservoir (Falter 1982).

## METHODS

Electrofishing surveys were conducted to track trends in size and relative abundance over time, assess the effects of timing on the results of these surveys, and explore new locations to expand surveys. These surveys were conducted during hours of darkness using an ETS MBS-1PD pulsator powered by a 5000s Honda generator. Two historical transects, and two exploratory transects were sampled. In 2015, the Dent transect was sampled on March 26, April 8, and April 27 (approximately one month before the historic timing), and an exploratory transect was conducted in the Canyon Creek arm on April 8 (Figure 39). In 2016, the Dent transect and an exploratory transect at Clear Springs were sampled on May 25, and the Magnus Bay transect was sampled on May 26, which coincides with the historic timing of these surveys (Figure 39). Attempts were made to collect all fish, of which species and total length (TL) was recorded. In addition, weight was recorded for all species in 2016. The second dorsal spine was collected from up to ten Smallmouth Bass from each 1-cm length bin for each transect.

Trends in indices of size and abundance were examined for surveys conducted during the historical timeframe (late May through early June). Catch-per-unit-effort (CPUE) was used as an index of relative abundance, and was calculated as the number of fish captured per hour of electrofishing. The CPUE of the age class in which bass achieved stock size (180 mm TL) was used as an index of recruitment. Proportional Size Distribution (PSD), formerly known as Proportional Stock Density (Anderson 1980; Nuemann et al. 2012), was calculated as:

$$PSD = \frac{N_q}{N_s} * 100$$

Where: $N_q$ = Number of fish ≥ quality length.
$N_s$ = Number of fish ≥ stock length.

Size indices for Smallmouth Bass are as follows: stock = 180 mm TL, quality = 280 mm TL, preferred = 350 mm TL, memorable = 430 mm TL, and trophy = 510 mm TL (Gablehouse 1984; Nuemann et al. 2012).

Body condition was evaluated using relative weights. Relative weight was calculated for bass obtained by electrofishing (spring), angling (spring and fall), and sampling tournaments (spring and fall). Relative weight ($W_r$) was calculated as:

$$W_r = \frac{W}{W_s} * 100$$

Where: $W$ = Measured body weight of an individual fish with a length of $L$.
$W_s$ = Standard weight of a bass with a length of $L$.

We used the $W_s$ equation for Smallmouth Bass proposed by Kolander et al. (1993), where:

$$W_s = 3.2 * \log_{10} L - 5.329$$

Age was estimated for bass by analyzing cross sections taken from the proximate end of the second dorsal spine. Dorsal spines were mounted in epoxy resin and sectioned with a Bueler Isomet low-speed saw. Sections were photographed using a Leica M80 stereoscope, with a TL5000 Ergo transmitted light base, and an IC80HD camera. Age was estimated by counting translucent rings, and for bass captured in the spring, the margin was counted as an annulus. Structures with defects, such as an eroded lumen, were not used in the analysis.

Growth was assessed by fitting the Beverton and Holt parameterization of the von Bertalanffy growth function to length at age data for bass captured during spring electrofishing, creel surveys conducted during the same month (May), and a tournament that was also sampled during May, 2016. The growth function, as follows, was fitted using the non-linear modeling function in JMP 9.0.0.

$$L_t = L_\infty\left(1 - e^{(-K*(t_c - t_0))}\right)$$

Where: $L_t$ = Length at age $t$.
$L_\infty$ = Maximum achievable length.
$K$ = Growth coefficient.
$t_c$ = Age at capture.
$t_0$ = Age at which length is zero.

For purposes of fitting the model, $L_\infty$ was held at held constant at the mean length of the oldest bass sampled in creel surveys or angler tournaments from 2014 to 2016. For comparison, a second von Bertalanfy model was fit to the mean length at age for 409 Smallmouth Bass populations across North America, using data from Beamesderfer and North (1995).

Exploitation rates were estimated using the IDFG “Tag, You’re It” program. Bass were collected during electrofishing surveys, at tournament weigh-ins, and from angling. A total of 692 bass were tagged with non-reward tags from 2013 to 2016; 135 in 2013, 189 in 2014, 62 in 2015, and 306 in 2016. In addition, 50 reward tags of $50 each were used to estimate the tag reporting rate specific to Dworshak Reservoir. To evaluate tag loss, we double-tagged 90 bass in 2015. Exploitation was calculated following the methods of Meyer and Schill (2014), using the number of returns reported in the first 365 days after release. Reporting rates for non-reward tags were calculated assuming complete reporting of reward tags following the methods of Meyer and Schill (2014).

The abundance of Smallmouth Bass ≥ 200 mm TL was estimated using a Lincoln estimator (Alisauskas et al. 2014). For this, we used the estimated harvest from a creel survey conducted from April 1 to August 31, 2016 (see Dworshak Reservoir creel surveys, this report). We also used the number of tags returned during this same period, and the reporting rate estimated from reward tags in 2016. The unbiased estimator, as given by Alisauskas et al. (2014), is as follows:

$$\hat{N} = \frac{(t+1)(\hat{H}+1)\lambda}{(r+1)} - 1$$

In this equation, $\hat{N}$ is the estimated abundance, $\hat{H}$ is the estimated harvest, $t$ is the number of tags at large during the creel survey, $r$ is the number of tags reported during the creel survey, and λ is the reporting rate. Since most bass were tagged prior to the creel survey, and several tags were reported prior, the number of tags at large was calculated as the number tagged minus the number reported prior to the creel survey adjusted by the reporting rate. The variance for this estimate was calculated as follows:

$$var(\hat{N}) = \left(\frac{t\hat{H}}{r}\right) \times var(\lambda) + \lambda^2 \times var\left(\frac{t\hat{H}}{r}\right)$$

Greater detail for calculating the variance be found in Alisauskas et al. (2014).

For comparative purposes, we re-estimated abundance for 2004 using data from Hand et al. (2008b) and assuming the same reporting rate as used for the 2016 estimate. For the original estimate, a reporting rate for Dworshak Reservoir was not available, so a range of rates from the literature was used (Hand et al. 2008b).

## RESULTS

The CPUE for Smallmouth Bass increased over time in 2015, whereas PSD peaked on the middle survey. A total of 94 Smallmouth Bass were captured along the Dent transect during three electrofishing surveys. Of these, 8 were captured on March 26 (214-455 mm TL), 21 on April 8, (202-360 mm TL), and 65 on April 27 (141-357 mm TL; Figure 40). An additional 11 bass were captured from the Canyon Creek arm on April 8. The water surface temperatures during these surveys were 6.6, 8.0, and 13.9°C, respectively. The CPUE increased steadily from 6.7 bass/h during the first survey to 45.8 bass/h on the last survey (Figure 41). The CPUE during the final survey was the higher than previously documented for the historic timing (Figure 42). During the April 8 survey, CPUE was 21.6 bass/h for the Dent transect and 66 bass/h for the Canyon Creek transect. The PSD increased from the first to the second sampling event, and then declined dramatically during the final event when smaller fish were sampled (Figure 41). However, PSD during the final sampling event was the higher than documented during the historic survey timing (Figure 43). During the April 8 survey, PSD was 86 for the Dent transect and 73 for Canyon Creek. Age estimated for bass captured in 2015 ranged from two to seven years (Table 5).

The CPUE for Smallmouth Bass from the Dent and Magnus Bay transects was higher than average in 2016, but PSD was lower than average. A total of 109 Smallmouth Bass were captured during electrofishing surveys conducted during May 2016 (historic timing). Of these, 53 were captured along the Dent transect (135-477 mm TL), 42 were captured along the Magnus Bay transect (135-371 mm TL; Figure 42), and 14 along the Clear Springs transect (162-434 mm TL). The CPUE was 43.1 bass/h for Dent and 34.2 bass/h for Magnus Bay, both of which were the second highest since 2004 (Figure 43). The CPUE for Clear Springs was 47.8 bass/h. The PSD for Dent was 4 and the PSD for Magnus was 15, both of which were the second lowest since 2004 (Figure 44). The PSD for Clear Springs was 17. Age estimated for bass captured in 2016 also ranged from two to seven years.

Recruitment of stock size Smallmouth Bass was lower than average for the Dent transect and higher than average for Magnus Bay. Smallmouth Bass consistently reached stock size (180 mm TL) at age-2 (mean TL = 189 mm). The CPUE for age-2 fish at Dent was 12 fish/h in 2015 and 35 fish/h in 2016. The CPUE of age-2 fish at Dent averaged 22.9 fish/h from 2004 to 2008. CPUE for age-2 fish at Magnus Bay was 19 fish/h in 2016. CPUE of age-2 fish at Magnus Bay averaged 5.9 fish/h from 2004 to 2008 (Figure 45).

Relative weight was a function of both TL and time of capture. The mean $W_r$ for Smallmouth Bass captured in the spring of 2016 (mean = 87) was lower than the mean $W_r$ for those captured during the fall (mean = 95). Mean $W_r$ decreased with increasing TL from approximately 200 to 300 mm, then increased with TL to around 350 mm, after which $W_r$ did not vary with TL (Figure 46). The mean $W_r$ was low compared to the $W_s$ for stock (mean = 77) and quality (mean = 75) Smallmouth Bass in the spring of 2016. The mean $W_r$ of stock (mean = 92) Smallmouth Bass increased by the fall of 2016, but was similar for quality (mean = 80) Smallmouth Bass (Figure). The mean $W_r$ was similar for preferred (mean = 99) and memorable/trophy (mean = 102) Smallmouth Bass in the spring of 2016. The mean $W_r$ increased for preferred (mean = 108) and both memorable and trophy (mean = 110) Smallmouth Bass in the spring of 2016 (Figure 3-9). The mean $W_r$ tended to be greater in the fall of 2015 than the fall of 2016 for all size classes.

Smallmouth Bass in Dworshak Reservoir grew at a rate similar to the North American average until they reached a TL of about 300 mm, then grew faster than the North American average. To fit the Von Bertalanffy growth model, $L_\infty$ was set to 543 mm TL, the mean length of the five oldest Smallmouth Bass (ages 9-10) encountered in any sampling during the past three years. The model predicts that the average Smallmouth Bass in Dworshak should grow to a memorable size (430 mm TL) by age-7, and grow to a trophy size (510 mm TL) by age-12 (Figure 48). The average Smallmouth Bass in North America should not reach a memorable size until age-10, and not reach a trophy size even if living to age-15 (Beamesderfer and North 1995).

The tag reporting rate estimate for Dworshak Reservoir was similar to the statewide reporting rate estimated by Meyer and Schill (2014) and no tag loss was detected. Of the 50 reward tags stocked in 2015, 21 were reported by anglers within two years of release. Assuming all reward tags were reported, the estimated report rate for non-reward tags was 51.0%, which is similar to the statewide reporting rate (54.5%) estimated by Meyer and Schill (2014). Of 90 bass that were double-tagged in 2015, 23 were reported by anglers, all of which had both tags present. Therefore, no adjustments were made for tag loss when estimating exploitation and abundance.

Exploitation rates were lower in recent years than in 2007, and lowest for the smallest and largest size categories of Smallmouth Bass (Figure 49). Mean annual exploitation for stock size and larger Smallmouth Bass (≥ 180 mm TL) was 15.9% from 2013 to 2016, compared to 21.9% in 2007 (Meyer and Schill 2014). Mean exploitation from 2013 to 2016 was 13.6% for stock Smallmouth Bass (180 to 279 mm TL), highest for quality (21.3%, 280 to 349 mm TL) and preferred (18.5% 350 to 429 mm TL) Smallmouth Bass, and lowest for memorable and trophy size Smallmouth Bass (7.1%, ≥ 430 mm TL; Figure 49).

Abundance estimates for 2004 and 2016 were similar. There were an estimated 42,000 (SE = 2,600 or 6%) Smallmouth Bass ≥ 200 mm TL in Dworshak Reservoir during 2016. By comparison, there were an estimated 45,000 (SE = 2,000 or 4%) Smallmouth Bass ≥ 200 mm TL during 2004.

## DISCUSSION

Dworshak Reservoir has steeply-sloped shorelines which are difficult to sample efficiently with typical boat electrofishing equipment. Because depth increases rapidly, only a narrow band along the shoreline is effectively sampled during most of these surveys. When surveyed during the historic timing (late May to early June) it is rare to encounter bass > 300 mm TL. When sampling earlier in the year, it appears that larger bass (> 300 mm TL) tend to move into shallower water earlier in the year, as water temperatures approach 8°C. Later, after water temperatures warmed, larger bass tend to move into deeper water, where they are less likely to be sampled, and smaller bass (< 300 mm TL) tend to move to shallow water in greater numbers. Therefore, the timing of these surveys is important in understanding the true size structure of the population. Surveys performed during the historical timeframe are likely to be useful for tracking trends in recruitment and estimating mortality of bass ≤ 300 mm TL. To sample larger bass (> 300 mm TL) more effectively with electrofishing equipment, surveys should be conducted earlier in the year when water temperatures are about 8°C. However, larger bass are still likely to be under represented with earlier electrofishing, which will bias mortality estimates high and PSD values low. Accurate mortality estimates are particularly important for evaluating the effectiveness of fishing regulations. Further research should be conducted to determine the most effective way to sample all size classes of bass in steep sided lakes and reservoirs, or incorporate data from multiple surveys reduce bias in estimates of population parameters in waters which are difficult to sample.

The abundance of Smallmouth Bass in Dworshak Reservoir appears to have been stable since 2003. The CPUE for all sizes of bass captured during electrofishing has been stable or increasing at our two trend sites. Furthermore, there is no evidence that abundance has declined between 2004 and 2016. Recruitment is also stable or increasing based on the increasing CPUE of age-2 bass observed at trend sites. Additionally, exploitation rates in recent years were at or below estimates from 2007 (Meyer et al) while the total fish harvested has increased (Hand et al. 2008a, Hand et al. 2008b, Wilson and Corsi 2016, creel survey section of this report). The analysis of reward tags used in 2015 indicates that reporting rates were the same as originally estimated in 2007, therefore exploitation estimates are likely comparable. However, creel methodologies have changed since 2004, and harvest estimates should be compared with caution. Even so, the available evidence is consistent with a stable or increasing population.

The PSD values for these surveys tended to be lower than the mean of the previous five surveys, suggesting that the size structure is shifting toward smaller fish. However, PSD values for electrofishing surveys are not representative of the population as a whole, as larger bass are underrepresented. Furthermore, the CPUE for quality fish remained similar to the mean of the previous surveys. Therefore, the decline in PSD is likely due to an increase in the abundance of smaller fish, not a reduction in larger fish. This is corroborated by the length distribution of harvested bass, where the mean size increased from 2015 to 2016 (Wilson and Corsi 2016, Section 4 of this report). Therefore, it appears that the density of larger fish was stable, but that densities of smaller fish increased.

The mean relative weight of bass in Dworshak Reservoir changed substantially as they grew. The mean $W_r$ of bass <350 mm TL was <80 in the spring, indicating poor body condition. Furthermore, the $W_r$ of these bass typically did reach 100 by fall, suggesting inadequate prey resources. While not measured, it is likely these bass had lower fat reserves, which could reduce over-winter survival. However, the mean $W_r$ for bass >350 mm TL was typically around 100 in the spring, and typically increased to >110 by fall, suggesting more than adequate prey resources. These fish likely had greater energy reserves, which may have resulted in higher over-winter survival (Oliver et al 1979). Although no recent data exists describing the diets of these fish, we speculate that bass begin to exploit kokanee *Oncorhynchus nerka*, the most abundant prey source for piscivorous fish in the reservoir, by the time they reach 300 mm TL, the size at which $W_r$ and growth increase. Further research is needed to better understand the relative importance of prey species and determine whether survival changes with size and prey availability.

In recent years, Smallmouth Bass >300 mm TL in Dworshak Reservoir grew quite fast compared to mean for North America (Beamesderfer and North 1995). Smallmouth Bass <300 mm TL, or about age-3, grew at a rate similar to the mean for North America. However, the growth rate increased substantially as Smallmouth Bass reached a TL of about 300 mm. This increase in growth rate concurrent with an increase in $W_r$, and is thought to occur when Smallmouth Bass are able to exploit kokanee as an abundant prey source, and may be the driver behind the production of memorable- and trophy-sized Smallmouth Bass that Dworshak Reservoir is known for, including two state records, and several others that were near the state record.

Angler exploitation for the bass population as a whole was relatively low (<20%) in recent years. Modelling performed by Beamesderfer and North (1995) predicted that exploitation rates in this range would not result in large declines in abundance and PSD for populations of average productivity. Indications are that the abundance and size structure of bass in Dworshak Reservoir has not been in decline in recent years. Lower exploitation rates for the smallest and largest size classes suggests that harvest-oriented anglers are more likely to release smaller fish, perhaps in order to keep larger fish as a part of their bag limit, but are less likely to catch the largest size classes (i.e. memorable and trophy). If this catch and release ethic amongst experienced bass anglers is maintained, then the potential effect of length restrictions will remain small.

## MANAGEMENT RECOMMENDATIONS

1. Maintain current fishing regulations for Dworshak Reservoir.

2. Investigate alternate methods to reduce bias in estimating Smallmouth Bass mortality rates so that current estimates can be confirmed or amended.

3. Investigate factors affecting growth rates in Smallmouth Bass to better understand how management actions affect the size structure of the population.

Table 5. Summary of results from electrofishing surveys conducted on Dworshak Reservoir in 2015 and 2016. Included are the date and location of the transect, as well as surface water temperature at the time of surveys conducted in 2015. Statistics include catch-per-unit-effort (CPUE, Smallmouth Bass/h), proportional size distribution (PSD), and annual survival rate (S).

| Year | Date | Location | Temp | CPUE | PSD | S |
|---|---|---|---|---|---|---|
| 2015 | March 26 | Dent | 6.6 | 6.7 | 38 | 42.9 |
| | April 8 | Dent | 8.0 | 21.6 | 86 | 19.7 |
| | April 8 | Canyon | 8.0 | 66.0 | 73 | 14.3 |
| | April 27 | Dent | 13.9 | 45.8 | 20 | 25.7 |
| 2016 | May 25 | Dent | | 43.1 | 4 | 30.5 |
| | May 25 | Clear | | 47.8 | 17 | 48.1 |
| | May 26 | Magnus | | 34.2 | 15 | 43.8 |

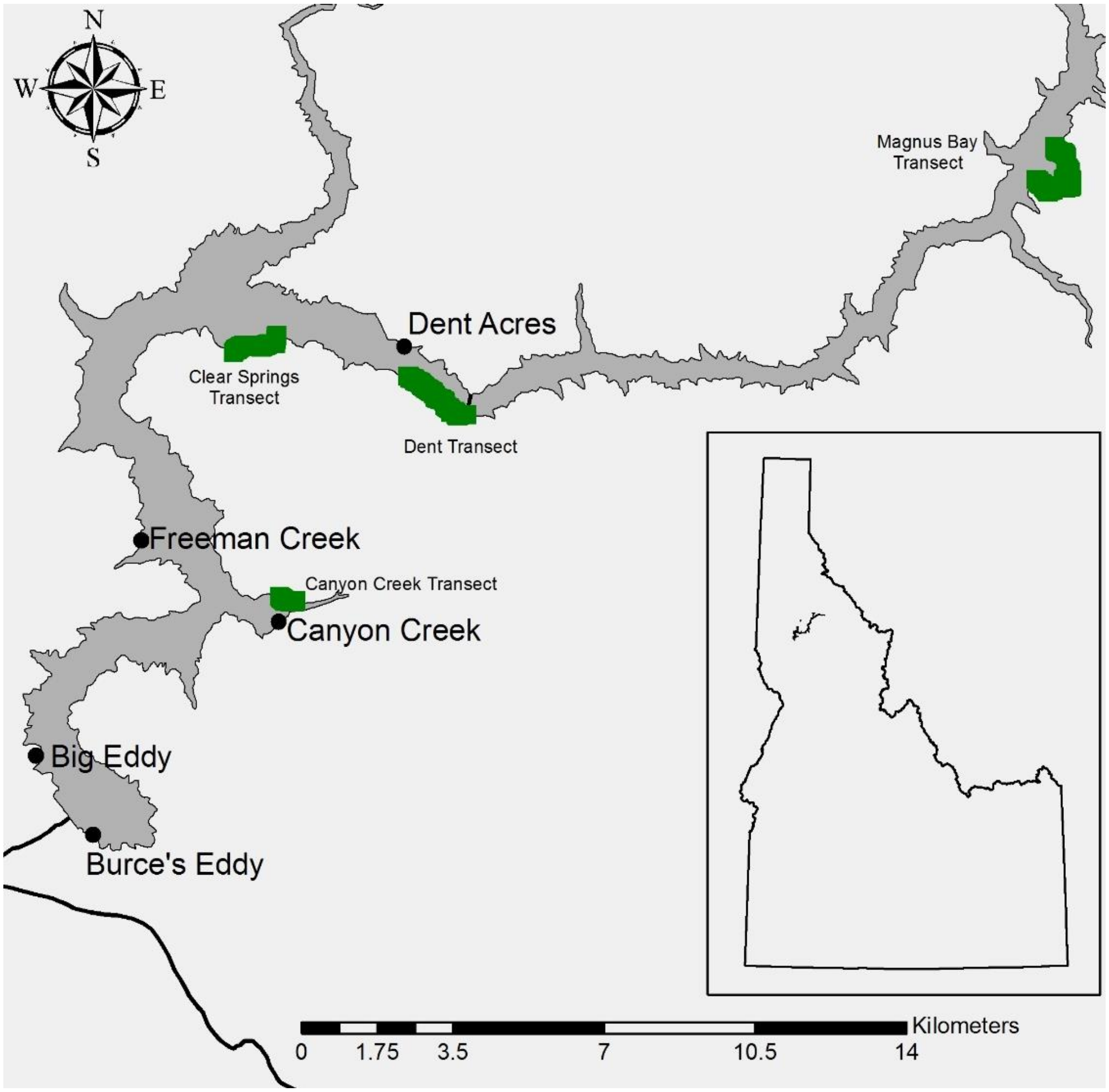


Figure 39. A map of Dworshak Reservoir with the locations of four electrofishing transects indicated by green swathes that were surveyed in 2015 and 2016.

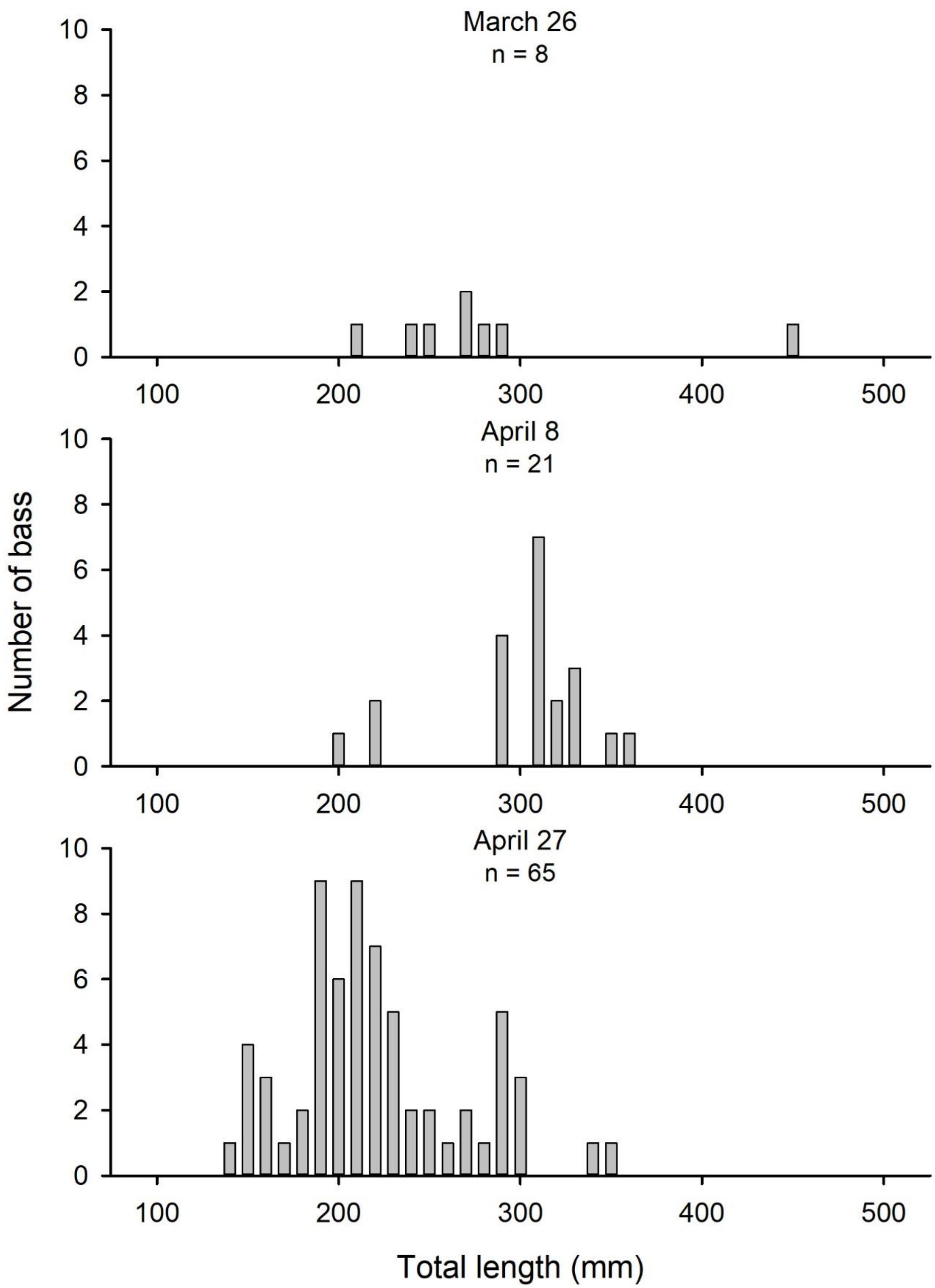


Figure 40. Length-frequency distributions of Smallmouth Bass captured while electrofishing the Dent transect on Dworshak Reservoir three successive times in 2015.

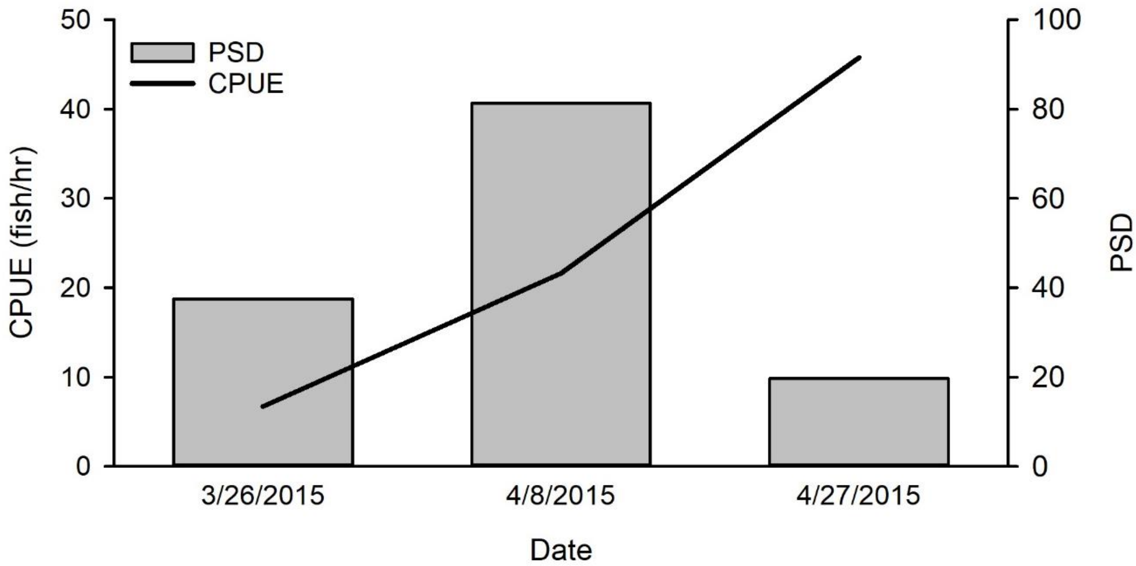


Figure 41. Catch per unit effort (CPUE, fish/h) and proportional size distribution (PSD) for Smallmouth Bass captured while electrofishing the Dent transect on Dworshak Reservoir three successive times in 2015.

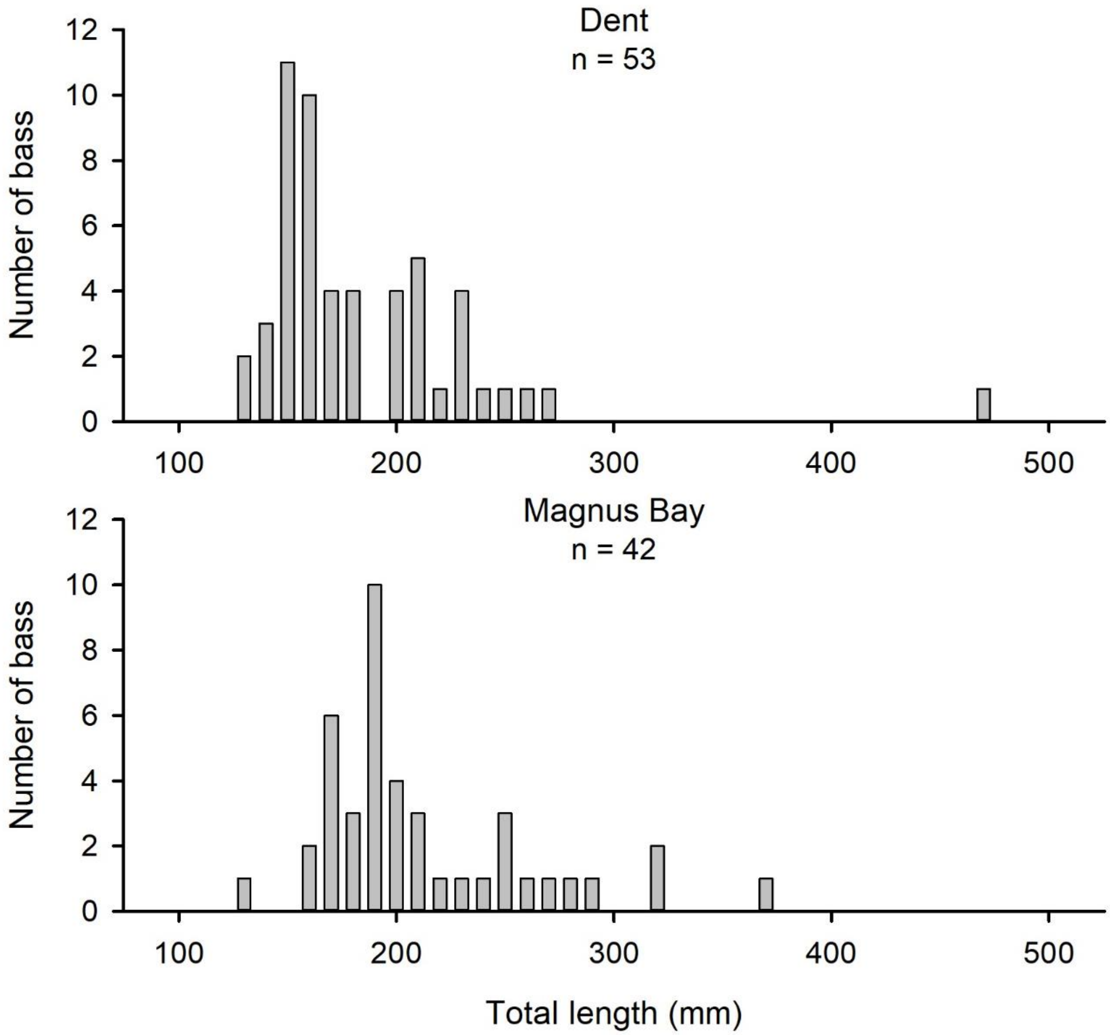


Figure 42. Length-frequency distributions of Smallmouth Bass captured while electrofishing the Dent and Magnus Bay transects on Dworshak Reservoir in 2016.

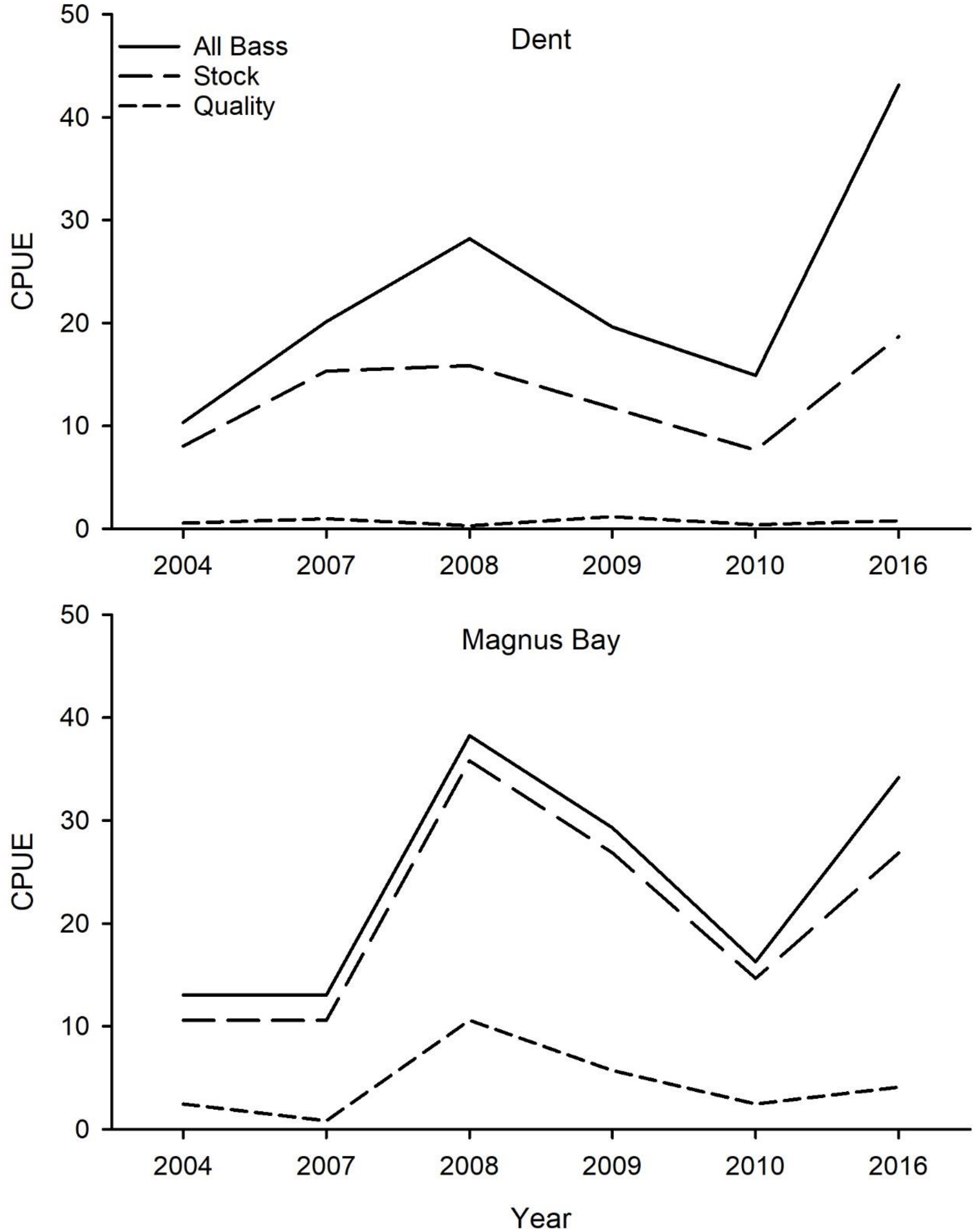


Figure 43. Catch per unit effort (CPUE, fish/hr) for three length categories of Smallmouth Bass captured while electrofishing the Dent transect on Dworshak Reservoir three successive times in 2015. Length categories include bass of all sizes, stock size or larger (≥ 180 mm TL), and quality size or larger (≥ 280 mm TL).

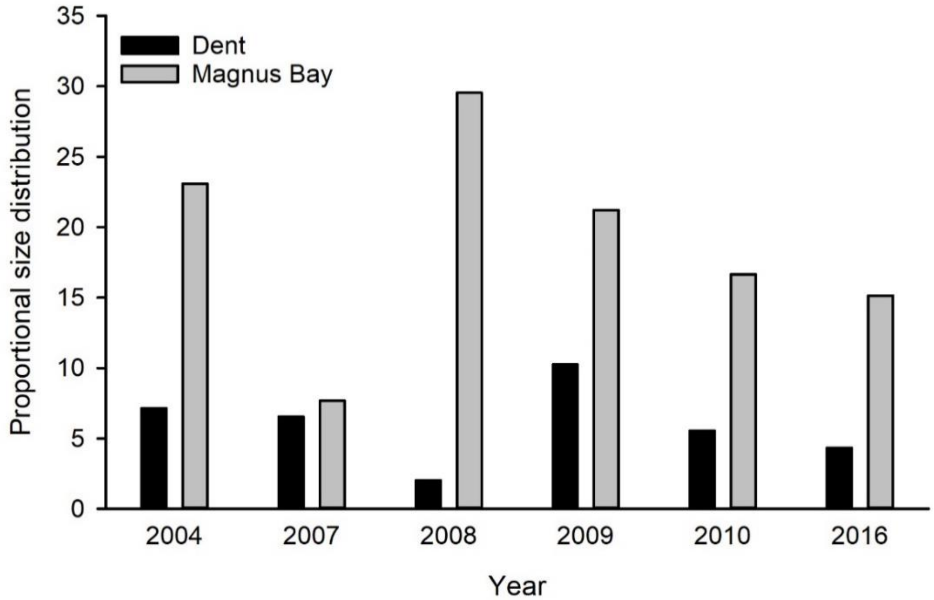


Figure 44. Proportional size distribution for Smallmouth Bass captured while electrofishing the Dent and Magnus Bay transects on Dworshak Reservoir from 2004 through 2016. Data are only shown for surveys conducted during the historic timeframe (late May to early June).

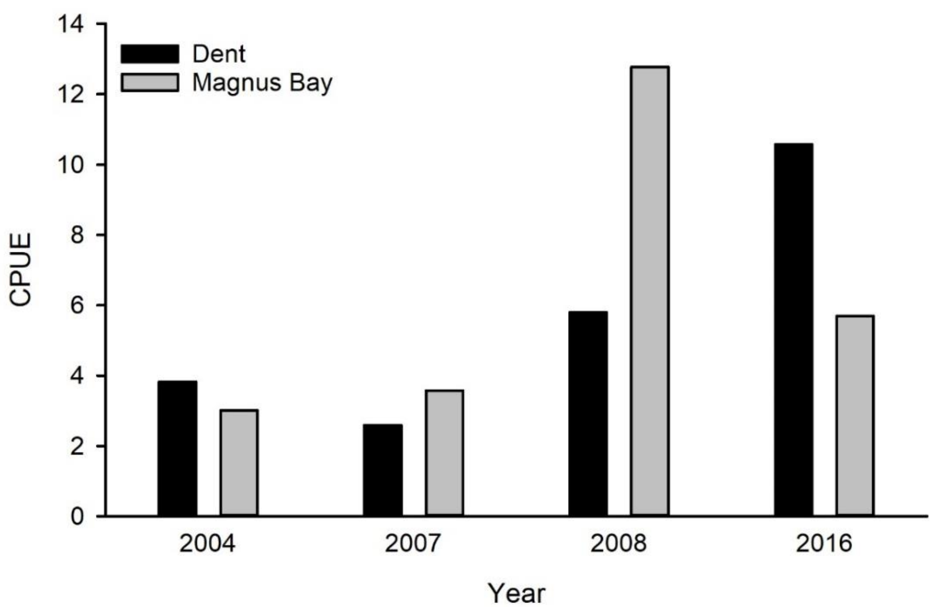


Figure 45. Catch per unit effort (CPUE, fish/h) for age-2 Smallmouth Bass captured while electrofishing the Dent and Magnus Bay transects on Dworshak Reservoir from 2004 to 2016. The length of age-2 bass ranged from 133 to 255 mm TL for the years shown.

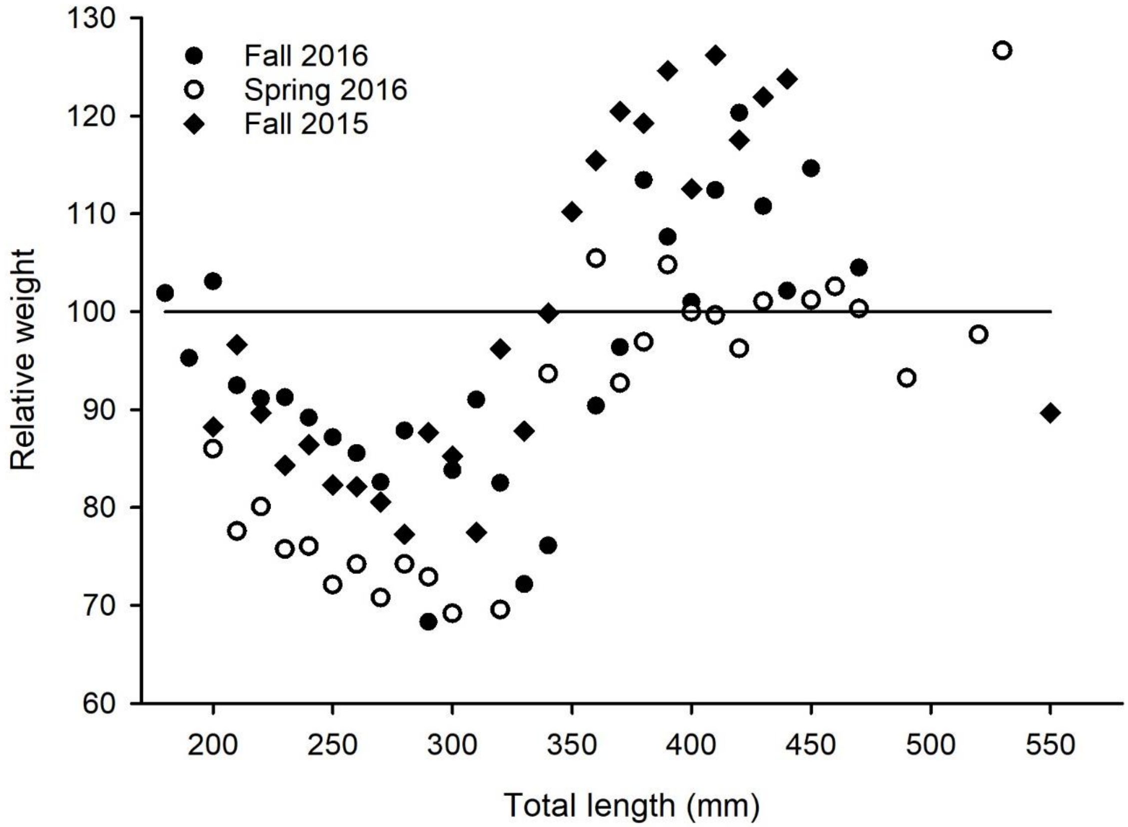


Figure 46. The mean relative weight of Smallmouth Bass sampled from Dworshak Reservoir for each 1 cm length bin, given by season (spring or fall) and year of capture (*n* = 335).

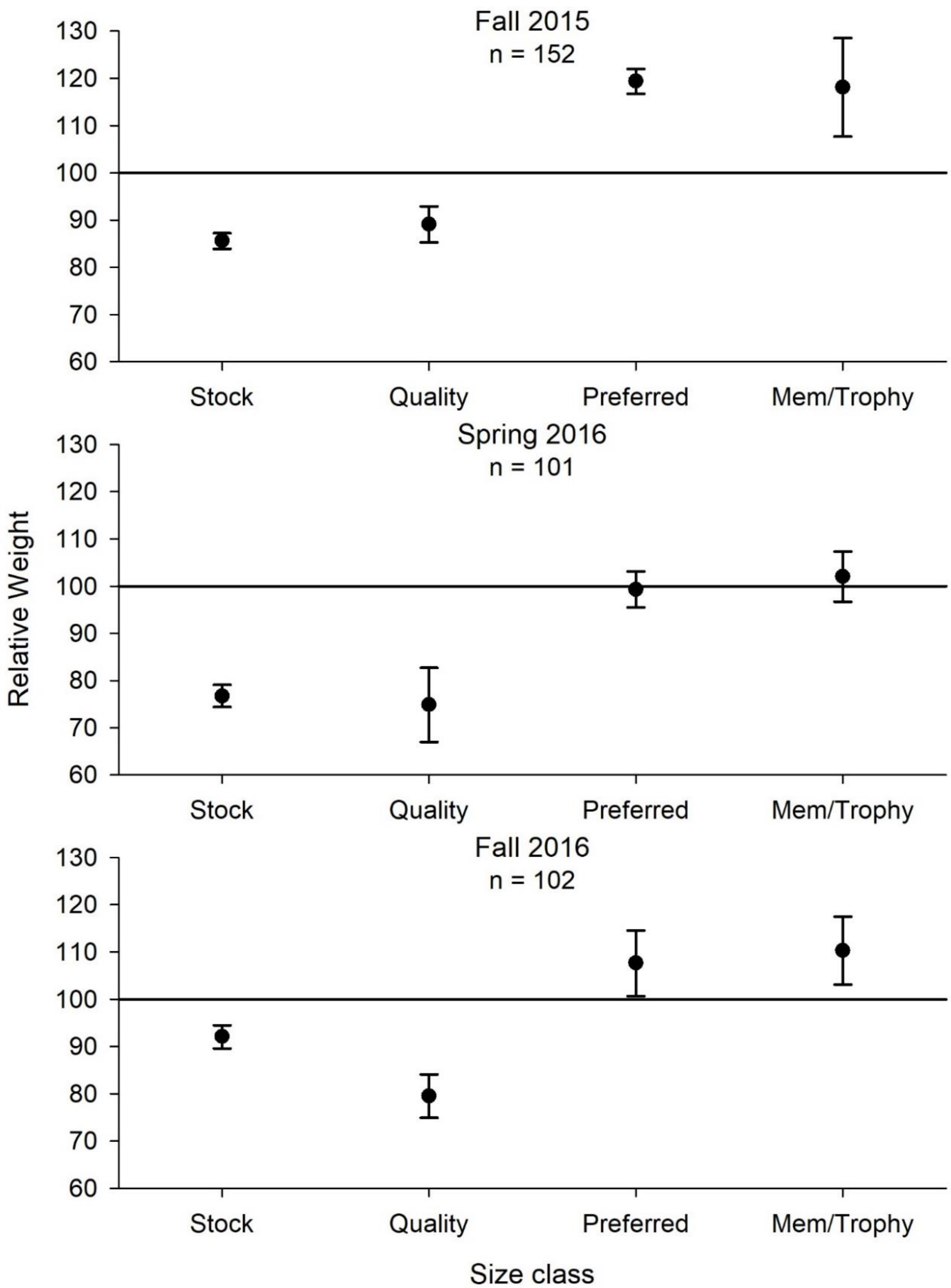


Figure 47. The mean relative weight for four size classes of Smallmouth Bass sampled from Dworshak Reservoir (*n* = 335). Means are reported by season (spring or fall), year of capture, and four size groupings; stock (180-279 mm TL), quality (280-349 mm TL), preferred (350-429 mm TL), and both memorable and trophy (Mem/Trophy, ≥ 430 mm TL). Error bars represent 95% confidence intervals.

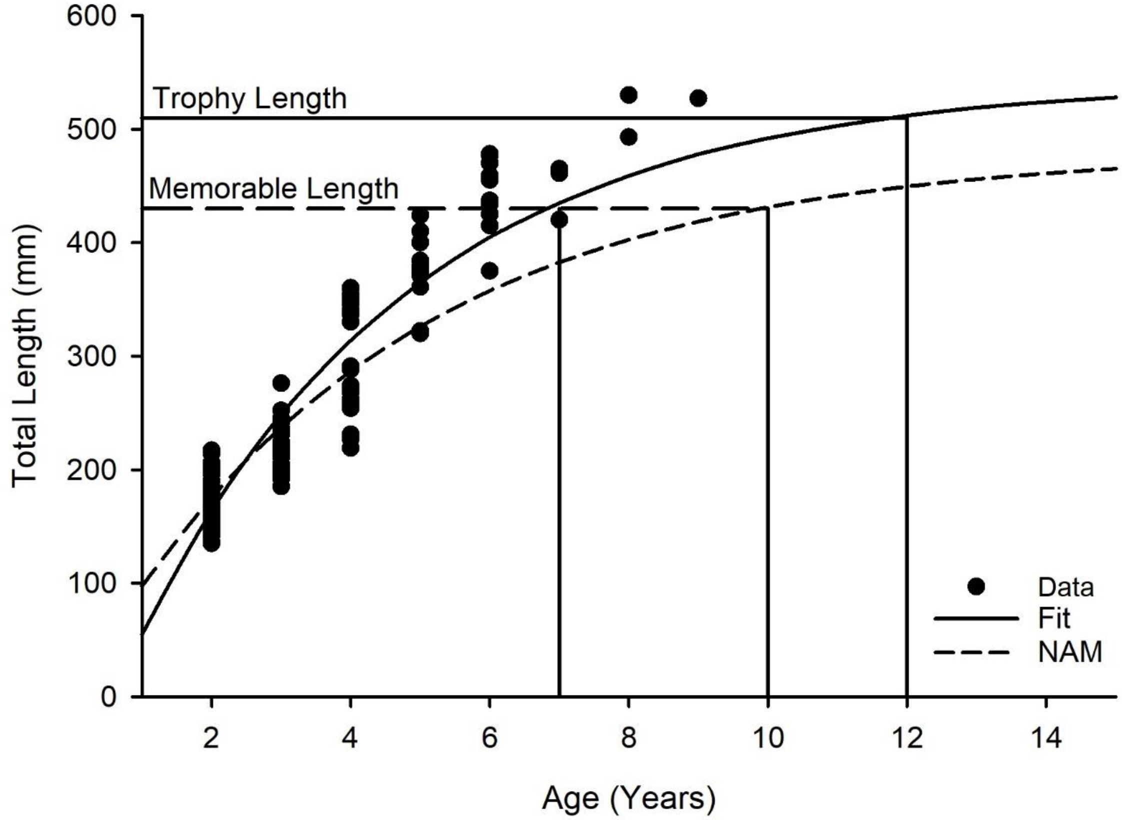


Figure 48. Von Bertalnffy growth curves for Smallmouth Bass sampled from Dworshak Reservoir in 2016, and the mean length at age for North America (NAM). Solids circles represent individual Smallmouth Bass (n = 116), the solid line represents the model fit to these data, and the broken line represents the NAM model. Horizontal lines depict the length at which Smallmouth Bass are considered memorable (430 mm TL) or trophy size (510 mm TL), and vertical lines depict the average age at which bass will reach these lengths for the Dworshak and NAM models.

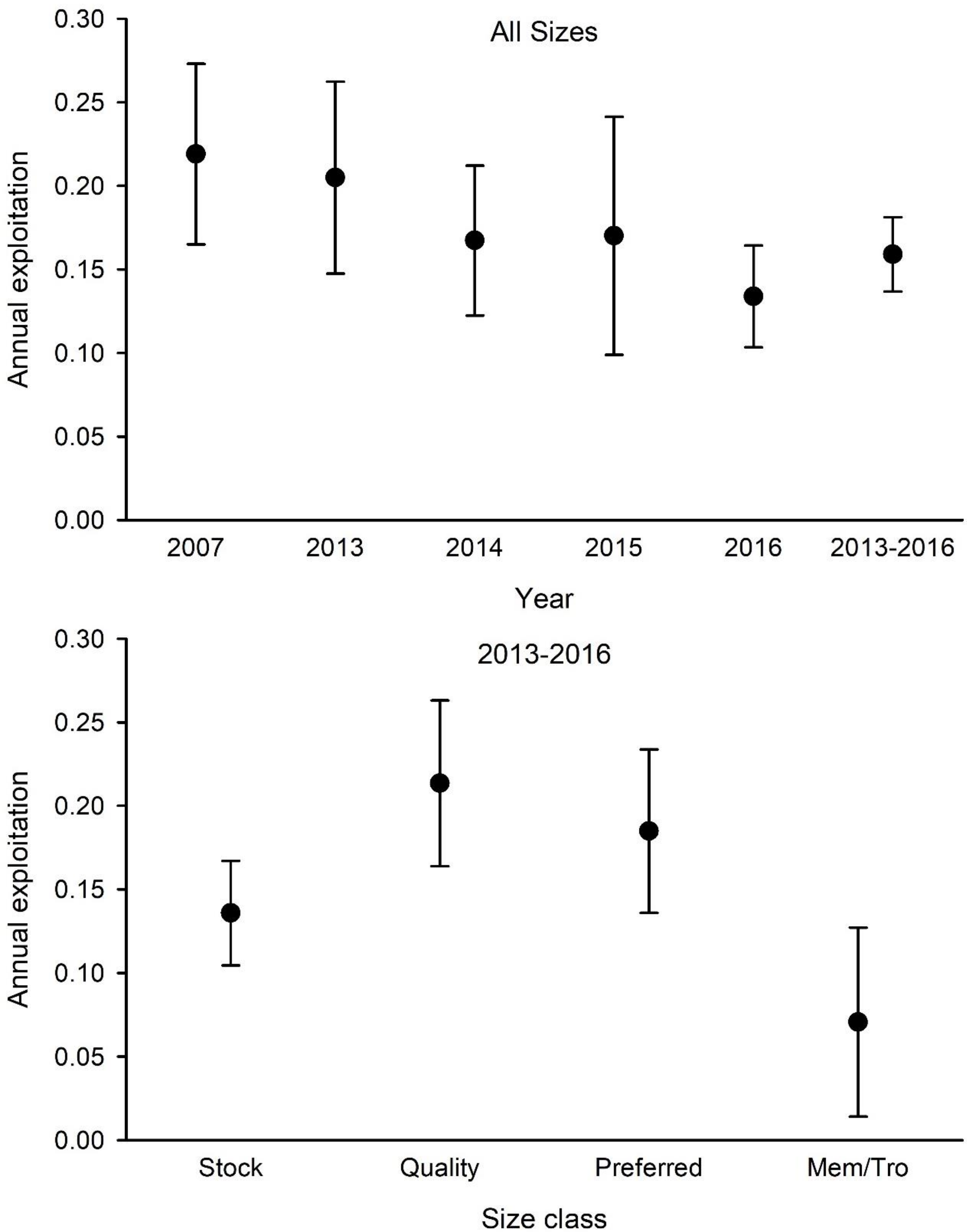


Figure 49. Exploitation rates and 90% confidence intervals for Smallmouth Bass in Dworshak Reservoir as estimated from tags returned by anglers. The top graph shows exploitation of the whole population by years, and for the period from 2013-2016. The bottom graph shows exploitation from 2013-2016 by four length classes; stock (180-279 mm TL), quality (280-349 mm TL), preferred (350-429 mm TL), and both memorable and trophy (Mem/Tro, ≥ 430 mm TL).

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## SOLDIER'S MEADOW RESERVOIR KOKANEE EVALUATION

## ABSTRACT

Kokanee *Oncorhynchus nerka* fry have been stocked annually in Soldiers Meadow Reservoir (SMR) since 2014 to establish a new fishery following a 2013 chemical renovation project. We have been evaluating this population through fall gill net surveys each year (2014-2016) to evaluate the growth and survival of both early and late spawning kokanee strains, and their potential for providing a fishery. Sampling in 2016 resulted in the collection of 264 kokanee. Kokanee averaged 280 mm (±4, 90% confidence interval) in length, a 12% increase from 2015 (251 mm, ±5), and 57% increase from 2014 (178 mm, ±5). Sampling in 2014 and 2015 indicated that average length for the early spawner strain was significantly larger than the late spawner strain. In addition, more individuals from the early spawner strain were captured each year. This suggested that both growth and survival were higher for early spawners. However, in 2016 this trend did not hold for the fish stocked in 2014. However, this is attributable to our sampling in late fall, which likely occurred after the mature (age 2+) early spawners had spawned and died. Thus, the fish we sampled in 2016 were a combination of the few remaining early spawners and the late spawners. At this point, evidence suggests that having a mix of both spawner strains could work well for SMR, with early spawners may provide a better fishery during spring-fall and the late spawners available for much of the ice fishing season. In the future, we recommend sampling earlier in the year in order to capture the early spawners before the spawn and die. This should improve our ability to compare the two strains. Additionally, we should analyze the cost-benefit of using both strains.

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## INTRODUCTION

Soldiers Meadow Reservoir (SMR) was renovated in 2013 with rotenone to remove stunted Yellow Perch *Perca flavescens*, Black Crappie *Pomoxis nigromaculatus*, and Black Bullhead *Ameiurus melas* populations (Hand et al. 2016). Following this management action, SMR has been primarily managed with put-grow-take kokanee *Oncorhynchus nerka* and put-and-take Rainbow Trout (RBT) *O. mykiss*. The decision to focus on a kokanee and RBT fishery was made based on preference indicated by anglers through email surveys and public meetings following the renovation (Hand et al. 2016). The current objectives are to evaluate SMR potential for providing a kokanee fishery, and to evaluate growth and survival of early versus late spawner kokanee strains.

## OBJECTIVES

1. Evaluate growth and survival of early versus late spawner kokanee.

2. Evaluate the potential of kokanee to provide a fishery in Soldier's Meadow Reservoir.

## STUDY AREA

Soldiers Meadow Reservoir is located approximately 45 km southeast of Lewiston Idaho, and 10 km west of Winchester, Idaho (Figure 1). It is a 47.8-ha reservoir with a mean depth of 5.6 m and a maximum depth of 14.0 m. Surface elevation is 1,378 m. Soldier's Meadow Reservoir was constructed for the Lewiston Orchards Irrigation District (LOID) to retain water for irrigation purposes. Its primary water supply is from Webb and Captain John creeks. Annual water level fluctuations of up to eight meters are common. Drawdowns usually begin by late June or early July as water is discharged for storage in Mann Lake. Low pool is generally reached by late fall towards the end of the irrigation season. Full pool is generally reached in May during spring runoff. Magnitude and timing of water level fluctuations is dependent on water yield in the LOID-managed watershed and irrigation demand. Facilities at this reservoir include primitive camping, boat ramp, and toilet.

## METHODS

Kokanee were sampled using overnight gill net sets (Hand et al. 2012) on October 19 - 20, 2016. This included two each of floating- and sinking-style monofilament gill nets 36.0-m long and 1.8-m high. The nets were divided into six equal size panels with bar mesh sizes of 10.0, 12.5, 18.5, 25.0, 33.0, and 38.0 mm. Monofilament diameter ranged from 0.15 to 0.20 mm. Gill nets were spread throughout the main body of the reservoir and placed in locations that were >2.0 m in depth that allowed for the net to be fully stretched out perpendicular to the shoreline (Figure 50). Data collected included net type (floating/sinking), fish species, lengths (total length, mm), weights (g), and otoliths were collected from approximately every third fish. Otoliths were collected using wire cutters to open the fish, and tweezers to locate and remove the otoliths. Each set of otoliths was stored in a coin envelope labelled with the date, reservoir name, and fish species, length, and weight. Catch-per-unit-effort (CPUE) and mean length of fish (mm), along with associated 90% confidence intervals, were calculated to compare with previous years. Significant differences in CPUE between years were determined to be those where 90% confidence intervals do not overlap.

Fingerling kokanee otoliths were thermally marked using different water temperatures while rearing at Cabinet Gorge Hatchery prior to stocking (Volk et al. 1990; Hagen et al. 1995). A different pattern was utilized for each strain and each year, which allowed for differentiation among the early and late spawner strains stocked in 2014 - 2016. Otoliths were collected from approximately every third fish for analysis. Collected otoliths were processed by technicians at the IDFG aging lab by mounting them in epoxy resin and sectioning with a Bueler Isomet low speed saw. Sections were observed under microscope to determine the thermal patterns on each set of otoliths. This allowed us to assign the appropriate strain and stocking year to each fish.

## RESULTS

In 2016, 7,000 early spawner (mean TL 76 mm) and 7,030 late spawner (mean TL 57 mm) kokanee were stocked into SMR (Table 6). Average length of early spawners (76 mm) was larger than late spawners (57 mm). This is to be expected, as early spawners were hatched first. Average length at stocking for each strain varied by year, although this is likely due to variation in stocking date.

The overnight gill net survey resulted in the collection of 278 fish, including 264 kokanee, 13 Rainbow Trout, and one Speckled Dace *Rhinichthys osculus*. Kokanee collected ranged in length from 137 to 329 mm (Figure 51), and averaged 280 mm (±4). This was a 12% increase in average length from 2015 (251 mm, ±5), and a 57% increase from 2014 (178 mm, ±5). Rainbow Trout collected ranged in length from 283 to 376 mm, and averaged 328 mm (±14; Figure 52). This resulted in a CPUE of 66 fish/net (±10) for kokanee and 4 fish/net (±2) for Rainbow Trout. For kokanee, this catch rate was higher than those observed in 2014 and 2015 (Table 7; Hand et al. 2018).

The mean TL of age-0 (164 mm) and age-1 (265 mm) kokanee collected in SMR in 2016 were larger than those sampled from numerous other Idaho and Washington reservoirs (Table 8). The kokanee captured in SMR also had a wider range of sizes of individuals caught compared to Dworshak Reservoir, probably due to the fact that both early and late spawner types were stocked into SMR (Wilson et al. 2013). These two stocks hatch at different times, causing late spawners to be smaller at stocking time.

Otoliths were collected from 87 kokanee, with thermal marks identified for 84 of those fish. Fish from all six mark groups (2014 - 2016, early and late spawners) were identified, including 2014 early spawners ($n$ = 3), 2014 late spawners ($n$ = 60), 2015 early spawners ($n$ = 16), 2015 late spawners ($n$ = 1), 2016 early spawners ($n$ = 3), 2016 late spawners ($n$ = 1). Average length between spawner type was similar for fish stocked in 2014 (Figure 53). Average length was larger for early spawners stocked in 2015 and 2016 (Figure 53). This is partly due to the differences in mean length at stocking. It must be noted that we did not effectively sample the entire population. Due to our late sampling date, the mature (age-2+) early spawners had likely already spawned and died.

## DISCUSSION

Kokanee fry have been stocked annually since 2014 to establish a new fishery following a 2013 renovation project (Hand et al. 2018). We have been evaluating this population through annual fall gill net surveys to evaluate growth and survival of both early and late spawner strains. The continued increase in average kokanee total length from 2014 to 2016 was expected, as the first kokanee were stocked in 2014, and they have had three growing seasons by the time gillnetting occurred in late October 2016 Figure 54). The average length of age-0 fish sampled in 2016 was 164 mm, a 17% increase over the 140 mm average in 2015. However, the 2016 data was based on only four fish. This increase in age-0 length was unexpected, as the increase in number and size of fish in the reservoir impacts the available food resources. Additionally, the fingerlings stocked in 2016 were smaller than those stocked in 2015 (Table 6). Like other fish species, kokanee growth and average length at age is generally density dependent (Reiman and Myers 1992; Walters and Post 1993). We would have expected a reduction in average size and annual growth over what was seen in previous years. Mean TL for age-1 kokanee did decline, with an average length of 229 mm in 2016 compared to 265 mm in 2015.

Part of our evaluation of the kokanee population was to determine if early or late spawner strains would grow and survive better in SMR. Sampling in 2014 and 2015 indicated the average length of the early spawner strain was significantly larger than late spawners (Figure 53). In addition, more individuals from the early spawner strain were captured during sampling in 2014 and 2015 (Hand et al. 2018b). This trend did not hold for sampling in 2016, as substantially more late spawners from the 2014 stocking were collected, and they had a larger average length (Figure 55). However, this is attributable to our sampling in late fall, which likely occurred after the mature (age 2+) early spawners had spawned and died. Thus, the fish we sampled in 2016 were a combination of the few remaining early spawners and the late spawners. Additionally, small sample size for several mark groups makes comparisons of length and growth difficult, at best. In the future sampling should be conducted earlier in the year to ensure a more representative sample. Additionally, otoliths should be collected from more kokanee to improve our sample size, and thus our ability to compare mark groups. We recommend collecting otoliths from at least 10 fish per centimeter length-group.

Evidence suggests that early spawners may provide a better fishery during spring-fall for anglers due their larger size at age-1 (Figure 55). However, with early spawners spawning and dying in the fall primarily at age-2, these fish are not available for the winter ice fishery. In contrast, the late spawners are available for much of the ice fishing season. Having a mix of both spawner strains could work well for SMR, with early spawners growing faster, and late spawners providing more winter ice fishing opportunity. In the future, we recommend sampling earlier in the year in order to capture the early spawners before the spawn and die. This should improve our ability to compare the two strains. Additionally, we should analyze the cost-benefit of using both strains.

Surveys over the next several years will be important to evaluate the success of these stockings and to determine what stocking densities are needed to maintain desired growth and catch rates. Kokanee generally have a maximum life span of 3 - 6 years (Rieman and Myers 1992). However, most kokanee populations in Idaho spawn at age 2 - 3 (Wahl et al. 2011; Janssen et al. 2012; Wilson et al. 2013). Sampling over the next few years will allow for an evaluation of the population at its full range of age classes and maximum density.

In addition to fish surveys, angler surveys should be conducted to evaluate angler effort, catch rates, and satisfaction. Determining when most of the effort is occurring will help us determine which strain would be more beneficial in SMR. Zooplankton sampling should also be conducted over the next few years to determine whether abundance and size is sufficient to allow kokanee to reach desired sizes (for anglers) before they reach sexual maturity (2 - 3 years old). After this time has elapsed, the success of the fishery can be evaluated and we can determine whether adjustments need to be made.

## MANAGEMENT RECOMMENDATIONS

1. Continue to sample kokanee and Rainbow Trout populations in 2017 and compare mean TL and CPUE of early vs. late kokanee. Conduct sampling in August to ensure capture of early spawner kokanee.

2. Collect otoliths from a minimum of 10 fish per centimeter length-group to improve sample size.

3. Evaluate zooplankton population.

4. Conduct angler survey to evaluate effort, catch rates, and satisfaction.

Table 6. Number of fish and average total length (mm) of kokanee stocked in Soldiers Meadow Reservoir, Idaho, from 2014 to 2016.

| Date stocked | Strain | Number stocked | Average length (mm) |
|---|---|---|---|
| 5/13/2014 | Early | 3,992 | 70.1 |
| | Late | 4,065 | 52.1 |
| 6/3/2015 | Early | 7,026 | 68.6 |
| | Late | 7,140 | 66.0 |
| 5/19/2016 | Early | 7,000 | 66.0 |
| | Late | 7,030 | 53.3 |

Table 7. Catch-per-unit-effort (fish/net), and associated 90% confidence intervals, of fishes collected during gill net surveys in Soldiers Meadow Reservoir, Idaho, from 2014 to 2016. Confidence intervals not calculated for 2014 due to small sample size.

| Species | 2014 | 2015 | 2016 |
|---|---|---|---|
| Kokanee | 21.0 | 46 (±8) | 66 (±10) |
| Rainbow Trout | 30.5 | 2 (±2) | 4 (±3) |
| Total | 51.5 | 48 (±8) | 70 (±11) |

Table 8. Comparison of kokanee average length at age in Idaho reservoirs.

| | | Length (mm) | | |
|---|---|---|---|---|
| Water Body | Survey Year | Age-0 | Age-1 | Age-2 |
| Lake Pond Orielle[a] | 2010 | 63 | 148 | 219 |
| Priest Lake[b] | 2010 | 40 | 180 | 265 |
| Cour D'Alene Lake[c] | 2011 | 40 | 110 | 170 |
| Deadwood Reservoir[d] | 2011 | <100 | 100-200 | 200-300 |
| Payette Lake[e] | 2011 | 45-58 | 105-133 | --- |
| Spirit Lake[c] | 2011 | 50 | 160 | 190 |
| Devil's Creek Reservoir[f] | 2012 | ~120 | ~280 | --- |
| Dworshak Reservoir[g] | 2013 | 84 | 222 | 270 |
| Soldier's Meadow | 2015 | 140 | 265 | --- |

[a]Wahl et al. (2011) [d]Butts et al. (2013) [f]Brimmer et al. (2013)
[b]Maiolie et al. (2011) [e]Janssen et al. (2012) [g]Wilson et al. (2013)
[c]Fredericks et al. (2013)


Figure 50. Location of four gill nets placed in Soldiers Meadow Reservoir, Idaho, on October 19, 2016.

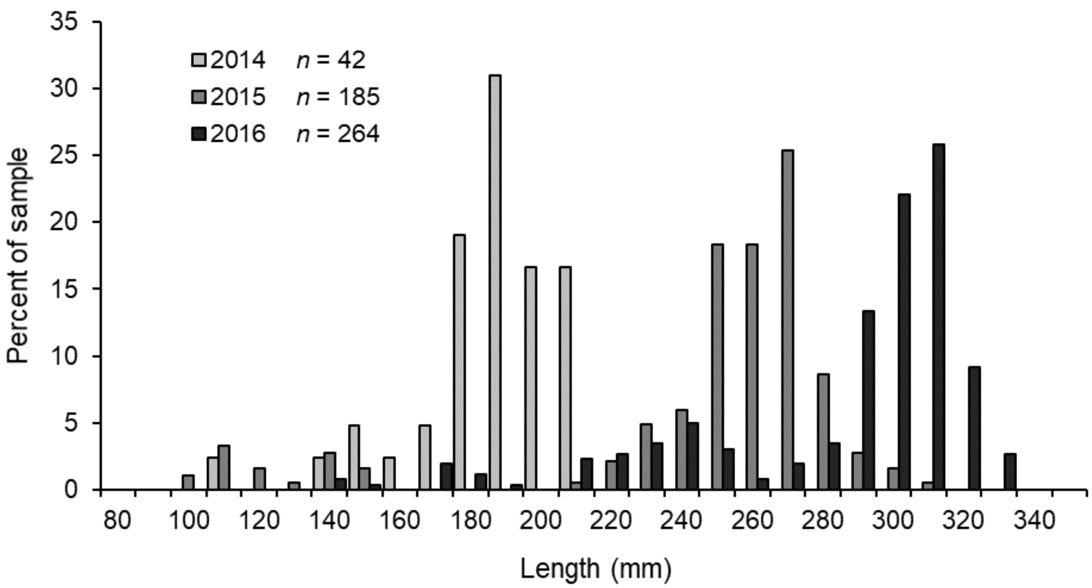


Figure 51. Comparison of length-frequency distributions of kokanee collected by gill nets in Soldiers Meadow Reservoir, Idaho, from 2014 to 2016.

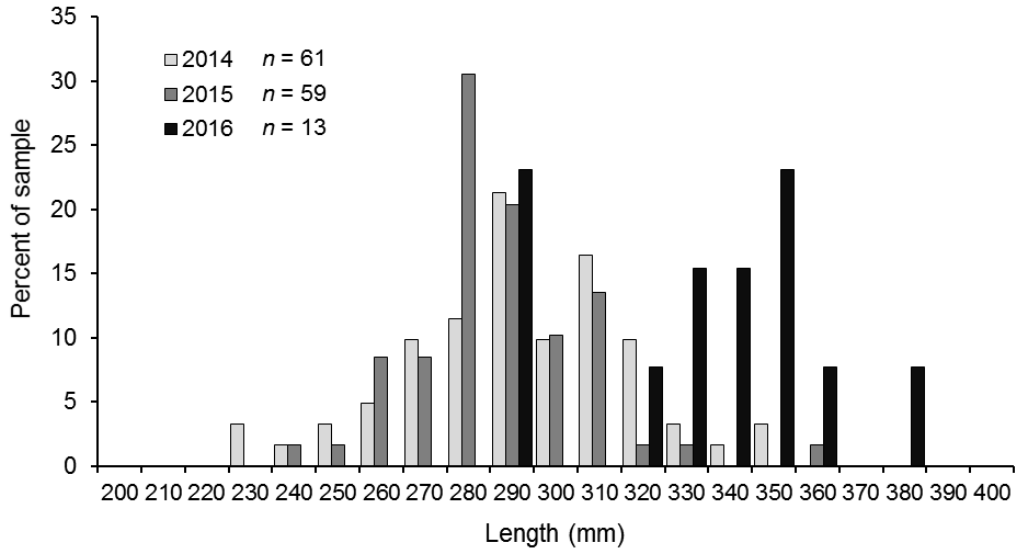


Figure 52. Comparison of length-frequency distributions of Rainbow Trout collected by gill net in Soldiers Meadow Reservoir, Idaho, from 2014 to 2016.

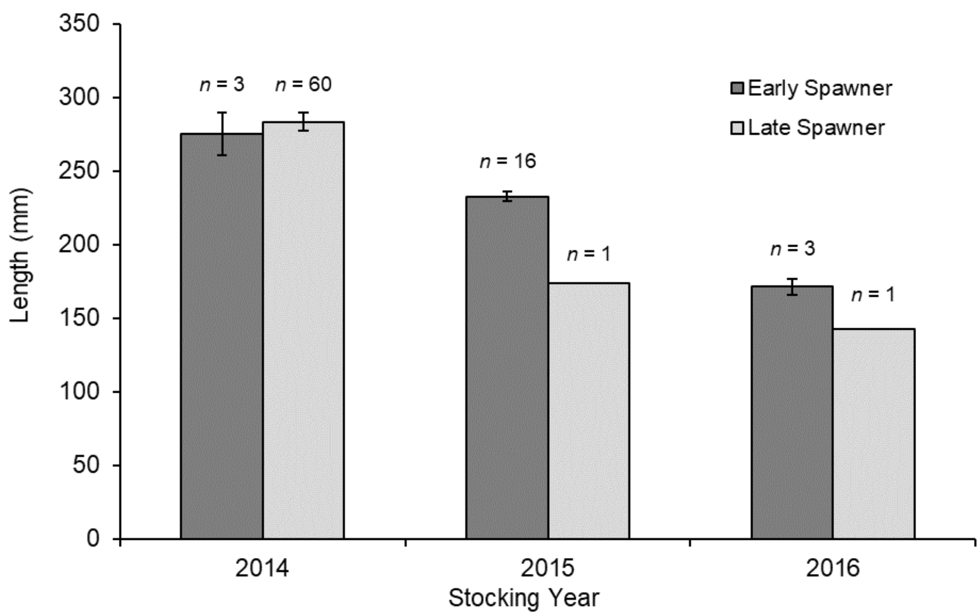


Figure 53. Comparison of average length (mm) at capture for kokanee collected by gill nets from Soldiers Meadow Reservoir, Idaho, in October 2016, by stocking year and spawner type. Error bars represent 90% confidence intervals.

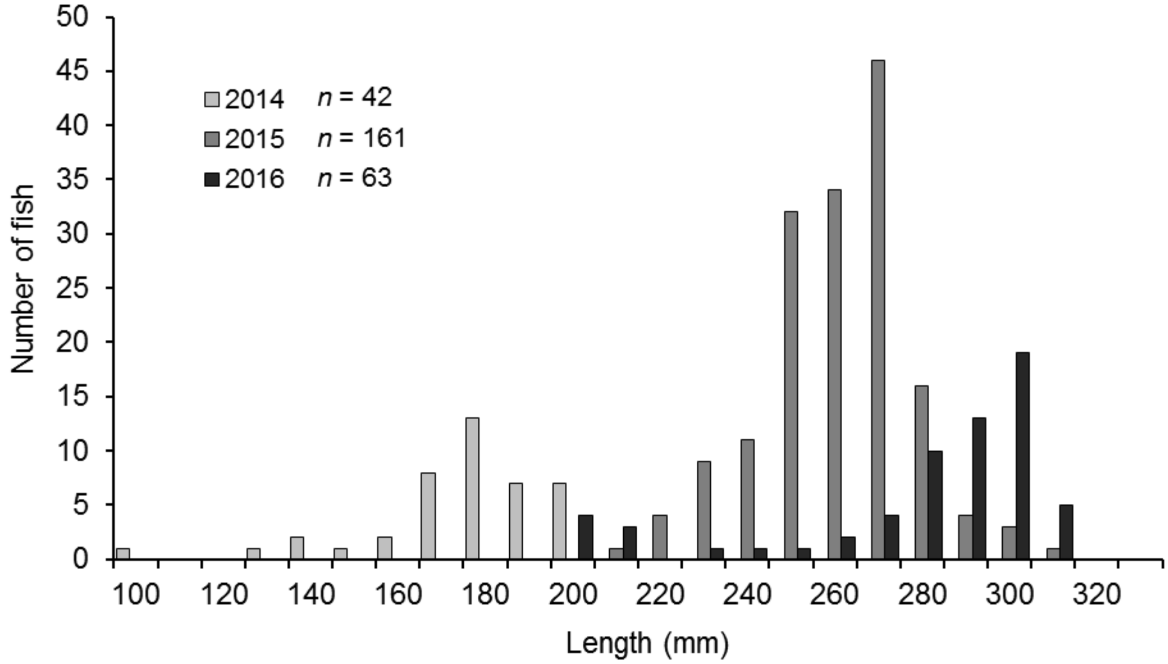


Figure 54. Comparison of length-frequency distributions of kokanee stocked in 2014 in Soldiers Meadow Reservoir, Idaho, based on October gill net sampling from 2014 to 2016.

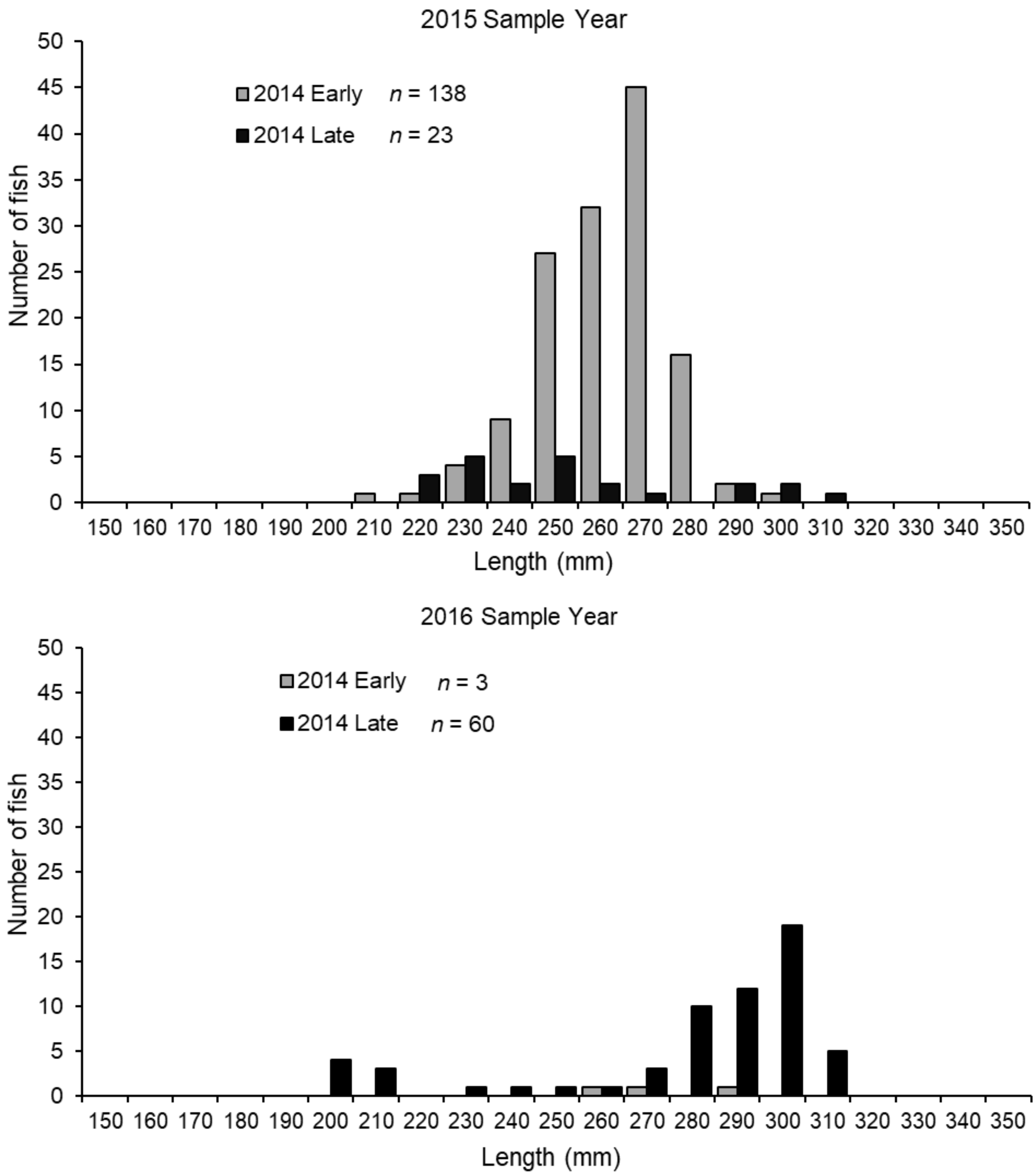


Figure 55. Comparison of length-frequency distributions between spawner stocks for kokanee stocked into Soldiers Meadow Reservoir in 2014, collected in October gill net surveys in 2015 and 2016.

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## SPRING VALLEY RESERVOIR: MONITORING THE EFFECTS OF RESERVOIR DRAWDOWN


### ABSTRACT

Loss of base flows during summer months is a primary factor influencing juvenile steelhead (*Oncorhynchus mykiss*) survival in the Potlatch River basin in Northern Idaho. The Spring Valley Reservoir (SVR) flow augmentation study evaluated the effects of using water releases from this headwater reservoir to benefit juvenile steelhead habitat downstream. We evaluated flow characteristics, water quality, and habitat condition in an 18-km reach downstream of SVR in relation to two different water release strategies in 2015 and 2016. In-stream residence time of the water released from SVR was at least 73 times faster when water was released while the downstream reach was still perennial (2016). We found that flow releases of 0.007 - 0.014 $m^3/s$ could maintain a perennial flow for 18 km downstream of SVR. Stream temperatures downstream of SVR did not exceed 17°C during the study, while control reaches on adjacent tributaries exceeded 22 °C or went dry. Dissolved oxygen levels at all sensor sites averaged 6.00 - 9.00 mg/L once the water release returned perennial flow versus an average of 2.07 mg/L prior to the water release. Prior to the flow augmentation in 2015, the upper 8 km had a 29% wetted length and the lower 8 km a 72% wetted length. The water release increased wetted length by 71% and 28%, respectively, for a 100% wetted length throughout the 18 km. In 2016, pool densities increased from 1.77 pools/100 $m^2$ to 2.22 pools/100 $m^2$, while the controls decreased from 0.61 to 0.17 pools/100 $m^2$. Within the reservoir, our primary concerns were potential impacts to the fishery and recreational use of the reservoir. This project resulted in a 1-m reduction in water surface elevation of SVR, and exposed banks up to 20 m in width, causing decreased accessibility. The high angle on the walkways to the docks, and lack of water near shore could cause access problems for many anglers. Our angler survey found no negative responses directly regarding the drawdown. Additionally, the drawdown does not appear to be impacting angler catch rates, as catch rates for warm-water fish and RBT were similar to pre-drawdown surveys. The only issue we observed in the fish survey was a decline in Largemouth Bass PSD during the drawdown years. However, with CPUE higher than in previous surveys, the low PSD appears to be primarily due to variable recruitment that is common in lowland lakes. The full effects of drawdown may not be evident for several more years and will need additional monitoring. If this water release becomes standard practice, we will need to be proactive to ensure continued access and use of the reservoir and facilities. In conclusion, an annual SVR 135,630 $m^3$ water release at 0.014 $m^3/s$ from July-September can create and maintain 18 km of perennial stream during low summer base flows with increases in habitat availability and pool abundance that meets flow, temperature and dissolved oxygen requirements for juvenile steelhead. However, improvements to both storage capacity and recreational access to SVR will be necessary before drawdown moves forward as an annual program. Potential alternatives will require further research to develop the best option(s) based on available funding.



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# INTRODUCTION

Changes in climate and anthropogenic influences are having adverse effects on ecological communities in many temperate waterways (Poff et al. 2006). In the Inland Pacific Northwest, Mediterranean climate patterns consisting of wet winters and dry summers dictate stream flow patterns with spring snowmelt accounting for of up to 75% of total runoff (Fritze et al. 2011). Spring runoff in mountain climates is occurring earlier in the water year, resulting in lower base flows and reduced habitat availability for aquatic assemblages (Hamlet et al. 2007). Anthropogenic impacts such as deforestation, agriculture, grazing, and urbanization have also greatly influenced stream flow (Wohl 2019). Anthropogenic changes have resulted in a decline in perennial systems and lower baseflows to watersheds of the Pacific Northwest (Roni et al. 2014)

Loss of perennial flow and decreased summer base flows have become a primary limiting factor for ESA-listed salmon and steelhead *Oncorhynchus mykiss* populations in many watersheds (Casagrande 2010, Ligon et al. 1995, Collier et al. 1996). Lower streamflow has been found to decrease habitat and food availability, negatively impacting fish production (Bjornn and Reiser 1991). In addition, declines in streamflow may result in higher water temperatures, embedded substrate composition, low dissolved oxygen (DO) levels, and restricted habitat connectivity (Poole and Berman 2001, Poff et al. 2006).

The preferred method for restoring summer streamflow to impacted streams is to focus on process-based habitat restoration efforts such as reconnecting floodplains and restoring riparian areas (Bohm 2007, Roni et al. 2015). Because the loss of summer streamflow has impacted ESA-listed fish such as steelhead, habitat restoration efforts have focused on this species in many watersheds (Roni et al. 2014). However, in areas where the problem is so extensive that restoration efforts become cost-prohibitive or changing climate is the cause (more winter precipitations as rain versus snow), other strategies must be explored. One strategy is to capture winter and spring runoff in reservoirs and then release water during summer to supplement base flows. A water release study conducted in a northern Idaho watershed found that releasing as little as 0.006 $m^3/s$ from a headwater storage reservoir restored perennial flow to 10 km of intermittent stream downstream (Sanchez-Murillo 2010; Treasure 2013; Brooks and Treasure 2014).

Our study area occurs in the Potlatch River watershed, which is unusual in that it is highly altered yet still contains a wild steelhead population with little or no genetic introgression from hatchery fish (Banks and Bowersox 2015). The Potlatch River likely has the strongest component of wild steelhead present within the Clearwater River Lower Main-stem population (Bowersox et al. 2008). A lack of summer base flow has been identified as a major limiting factor for steelhead production in the lower Potlatch River drainage (Banks and Bowersox 2015). Because of the high potential to increase steelhead production, this watershed is a high priority for restoration projects that typically include riparian and floodplain restoration, instream habitat improvements and barrier removals. The lower Potlatch River drainage is an example where watershed-level anthropogenic effects are to the extent such that restoration efforts focused on fixing the root of the problem (historical perennial flow conditions) using restoration methods that mimic natural processes may be cost-prohibitive or take multiple decades to exhibit a biologically meaningful response. As such, more immediate strategies to restore summer flows were considered.

In 2015, IDFG initiated a pilot project to evaluate the effectiveness of releasing water from Spring Valley Reservoir (SVR) to benefit steelhead rearing in the 18 km of stream directly downstream of the reservoir. This reach includes Spring Valley Creek (SVC) and Little Bear Creek

(LBC). Juvenile steelhead production in both streams is limited by habitat and density-dependent effects (Uthe et al. 2017). This project was designed to use water releases from SVR to maintain perennial flow in SVC from late summer through fall.

This water release strategy results in a larger than normal reduction in surface elevation of SVR. The water level in SVR is managed through using dam boards fitted on the spillway, which are installed each spring after peak run-off in order to capture some of this water and increase the maximum level of the reservoir during the peak recreation season (spring-summer). During 2015, the total water volume released from the reservoir was estimated at 133,092 $m^3$. This equated to a 0.61-m reduction in surface elevation. At the end of the water release project, the reservoir surface elevation was 1.07 m below full pool, indicating that approximately 0.46 m of the reduction was due to evaporation and seepage. In 2016, the water release from SVR occurred from June 6th to October 10th, and began while perennial flow was still present in lower SVC. The water release plan for 2016 was modified from 2015 to identify the minimum amount of water needed to be released from the SVR to maintain perennial flow in the stream downstream of the reservoir.

Monitoring the effects of the drawdown project at SVR will be important for the future management of the reservoir fishery. Spring Valley Reservoir is an important fishery in the Clearwater Region's lowland lake and reservoir program given its close proximity to Moscow, ID and Pullman, WA. Spring Valley Reservoir is the closest public fishery to both of these communities and therefore receives high levels of angler effort. An economic survey conducted in 2011 estimated anglers took 10,507 trips to SVR for an estimated total economic expenditure of $382,791 during a one-year period (IDFG, *unpublished data*). If the water releases into Spring Valley Creek prove successful at increasing steelhead production, this program may be implemented on a long-term basis resulting in annual drawdown in SVR. The biggest concerns for SVR in relation to this flow enhancement project are the potential impacts on the fish population and angler satisfaction. A declining fishery or reduced angler satisfaction at SVR would likely reduce recreational usage (including angling) of the reservoir. If this program continues, information collected from this and follow up assessments will help us understand any impacts that are occurring and strategies that can be used to maintain high levels of angler satisfaction and a desirable fishery.

In 2016, a second year of the pilot study was conducted. Water releases from SVR began during June before the test reach went intermittent to identify the minimum flow needed to maintain perennial flows downstream. Stream flow, water quality, and habitat condition was monitored within the study reach below SVR. In addition, due to its recreational and economic importance, we conducted fish population and angler surveys of SVR to monitor potential impacts to the fishery and angler satisfaction during the course of the drawdown project.

## OBJECTIVES

1. Evaluate the effectiveness of releasing water from Spring Valley Reservoir to benefit steelhead rearing habitat in an 18-km reach downstream of the reservoir.

2. Evaluate the effects of water level drawdown on fish populations, angler use, and angler satisfaction at Spring Valley Reservoir.

## STUDY AREA

The Potlatch River basin is a complex landscape in northern Idaho encompassing 152,622 ha. The main-stem Potlatch River is approximately 89.4-km long and is the largest tributary to the lower Clearwater River. The Potlatch River enters the Clearwater River about 20 km east of Lewiston, Idaho. Elevations range from 244 m at its mouth to 1,000 m in the headwaters. Within the Potlatch River watershed is the Little Bear Creek (LBC) watershed including SVR and its downstream tributaries in which the study occurred (Figure 56). These tributaries are SVC, which travels 8 km from SVR before it merges with Nora Creek and becomes LBC which flows another 8 km to its mouth where it merges with West Fork Little Bear Creek (WFLBC), the lowest extent of our study. From this point LBC continues another 7 km before merging with Big Bear Creek (BBC), the other primary watershed of the Big Bear basin. Big Bear Creek then flows an additional 1.16 km before joining the Potlatch River.

The 32,000-ha BBC watershed is characterized by steep basaltic canyons rimmed by rolling cropland. The predominant stream type in the lower watershed is a canyon stream characterized by high gradient, large substrate size, and riffle/pocket water habitat. Land ownership in the watershed is primarily private and principal land uses are agriculture (62%) and forest (35%) production (Hortness and Berenbrock 2001). Mean annual precipitation within the watershed is 72 cm which typically comes as winter snowpack and spring and fall precipitation.

Spring Valley Reservoir is a 19.8-ha reservoir located in Latah County approximately 13 km east of Troy, Idaho at an elevation of 726 meters (Figure 1). It is approximately 29 km from Moscow, ID (pop. 24,080) and 44 km from Pullman, WA (pop. 29,913). It has a mean depth of 3.6 m, a maximum depth of 8.8 m, and a maximum volume of 906,600 $m^3$. The reservoir is eutrophic and prone to algal blooms in the late summer. The surrounding watershed is dominated by timberlands with some limited agricultural areas above the reservoir. Spring Valley Reservoir was originally constructed in 1961 by IDFG to create a recreational fishery. In 1993, the spillway was reconfigured to meet the dam safety specifications of the Idaho Department of Water Resources. Facilities at the reservoir include a boat ramp, picnic pavilion, vault toilets, numerous ADA-accessible fishing docks, and primitive camp sites. Most inflow into the reservoir occurs during spring snowmelt, usually from late March to early May. By the end of the summer, the surface elevation of SVR generally decreases ~0.6 m due to evaporation and seepage.

## METHODS

### Water release evaluation

We evaluated the response of habitat conditions to flow supplementation with a Before-After-Control-Impact (BACI) design. This design allowed assessment of treatment effects, habitat response to the treatment, and magnitude of effects. The treatment reach encompassed SVC and LBC and covered 18 km downstream of the reservoir (Figure 56). Control reaches were located in WFLBC in 2015 (one control site) and WFLBC and BBC in 2016 (four control sites). Parameters of interest included streamflow, wetted habitat, pool density, water temperature, and dissolved oxygen. We assessed wetted habitat and pool density during habitat surveys in the treatment reach prior to the flow supplementation and throughout the treatment period. Water temperature, depth, and dissolved oxygen (DO; mg/L) were monitored via remote, continuous data loggers (HOBO model U26; Onset Computer Corporation) at stations located every 2 km throughout the treatment reach (Figure 56). Treatment data were stratified by flume, meadow (0

- 8 km), and canyon (8 - 18 km) reaches to assess local effects. We used graphical comparisons for inference.

Two water release strategies were evaluated during the two-year study period Figure 57). In2015, releases from SVR began on August 3 after perennial flow had ceased and ended on October 21 once fall flows returned. In 2016, water releases began on June 6, before perennial flow ceased, and ended October 10. Our goal in 2016 was to release enough water to maintain flows of at least 0.006 $m^3/s$ at monitoring sites at the meadow and canyon interface (SVC8 and LBC10) for the duration of summer.

A Parshall 15.24-cm flume with 1.82-m sealed wing walls was installed approximately 50 m below the outlet pipe to capture the entire volume of water being released from the reservoir. We referenced the Parshall flume discharge table to estimate flows (USBOR 2001). Whenever water releases were modified during the study, flow measurements occurred 3-6 hours after valve adjustments to allow outflow to stabilize.

Streamflow was measured at each of the treatment and control sites using Decagon CTD-10 pressure sensors and Decagon EM50 data loggers. The sensors and loggers were inserted into a perforated PVC protective housing and deployed during the month of May in 2015 and 2016. The sensors were deployed in pools (at least 30 cm deep) immediately adjacent to each two-km monitoring station. The pressure sensors were downloaded at least once monthly using Decagon ECH2O software version 1.74.

A Marsh-McBirney model 2000 portable flow meter was used to measure and calculate stream velocities and discharge where each of the Decagon sensors were deployed. Readings occurred any time water releases from the reservoir changed (six in 2015 and five in 2016) and once in June and July prior to water release to calibrate the flume with the flow meter. At each site, stream depth and water velocities were taken at a minimum of 15 locations across the channel. The flow sensor was set at 60% total depth for thirty seconds before a velocity was recorded. Total discharge at each site was calculated in cubic meters per second ($m^3/s$) based on wetted area and water velocity. Discharge data were paired with pressure sensor data to build a rating curve for estimating flows at various water depths for each site. This allowed us to calculate streamflow on an hourly basis for each sample site for the duration of the study.

Stream temperature (°C) was recorded hourly using the Decagon CTD-10 pressure sensor and Decagon EM50 data loggers deployed at each site. Mean daily temperatures were calculated for each site for the duration of the study.

In 2015, DO was monitored at SVC0, SVC2, SVC6, SVC8, LBC10, and control site WFLBC. In 2016, DO was monitored at SVC2, LBC10, LBC16, and two control sites (WFLBC and Middle BBC). The change in DO monitoring sites in 2016 was made to look at the influence further downstream due to the results witnessed in 2015 (i.e. anoxic water at SVC0 was fully oxygenated by the time it reached SVC2). Onset HOBO U-26 Dissolved Oxygen Loggers measured DO at 30 minute intervals. HOBO loggers were deployed in the same pools as pressure sensors. We summarized instantaneous DO into mean daily values at each site to monitor daily change during the treatment period. The loggers were cleaned and data were downloaded every two to three weeks.

Habitat surveys were conducted from July 22 to September 23 and from July 18 to August 30 in 2015 and 2016, respectively. Habitat surveys were conducted at all sites during the same general time period when flows stabilized. Surveys were conducted at a broad range of water release levels to provide a wide variety of conditions to evaluate each discharge level. In 2015 and 2016, 200-m habitat surveys were conducted at every 2 km station except SVC 0, where wetted pool habitat was assumed to exist at the water release location. Habitat surveys occurred 100 m above and below sensor and logger locations for a total of 200 m surveyed at each water quality monitoring site. Measures of total wetted habitat, pool abundance, and pool quality were collected within each transect. Pools were identified using the Low Water Habitat Availability Protocol (Bowersox et al. 2008, Uthe et al. 2017), where pools were enumerated, maximum depth, modal depth, pool width and length were used to quantify pool area and volume.

**Reservoir evaluation**

Angler creel and opinion surveys consisted of interviews conducted during six days from August 24 to October 30, 2016. This included three week days and three weekend days. Surveys were conducted when scheduling allowed, and were not randomly selected or stratified and were only used to evaluate catch rates for the various fish species caught. Creel clerks parked at the main access point to the reservoir from 9am to 5pm and all anglers and non-anglers leaving the lake during were interviewed to collect completed trip data. Total hours fished, number of fish caught, fish species, and fish lengths (total length, mm) were recorded during interviews. Each angler was interviewed separately and not as a group. Angler opinion surveys were also conducted in conjunction with the creel surveys (Table 9). The catch rates and angler opinions were then compared to other years to see if differences occurred. With our limited sampling, we were unable to evaluate any potential influence on angler effort.

To evaluate the fish community in SVR, an electrofishing survey was conducted on May 17, 2016. Boat mounted electrofishing was performed using pulsed DC current from a Honda EU7000iAT1 generator and an Midwest Lakes Electrofishing Systems (MLES) Infinity pulsator. Four 10-minute electrofishing subsamples were conducted on the reservoir for a total of 2,400 seconds of electrofishing effort. Only four subsamples were conducted due to poor weather conditions which prevented us from completing the six planned 10-minute subsamples. Electrofishing was conducted along the shoreline in a clockwise direction, with each subsample started at the locations marked in Figure 58. The survey was conducted at night, and we attempted to net all fish observed. Species, total length (mm), and weight (g) were recorded for each fish sampled. We compared catch rates and sizes of fish to previous years to assess potential effects of the water level drawdown. Catch-per-unit-effort (CPUE; fish/h) and weight-per-unit-effort (WPUE; kg/h) with associated ± 90% confidence intervals were calculated for total catch and each species to compare with previous years. Significant differences between years were determined to be those where 90% confidence intervals do not overlap. Mean length of fish (± 90% confidence intervals) were compared by species between years using a standard two-sample *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.1$. Confidence intervals could not be calculated for surveys prior to 2010, as those surveys were not divided into subsamples. Weight-per-unit-effort was not calculated for surveys prior to 2001 due to lack of individual fish data.

Proportional Size Distribution (PSD) was calculated for Largemouth Bass (LMB) *Micropterus salmoides* and Bluegill (BG) *Lepomis macrochirus* to provide information on population size structure using the following formula (Guy et al. 2007; Neumann et al. 2012):

$$PSD = \frac{\# fish \geq qualitysize}{\# fish \geq stocksize} * 100$$

Quality size and stock size correspond to lengths considered to be the minimum size at which anglers will first catch the species (stock) and consider the fish to be of desirable size (quality). These lengths are 200 and 300 mm for LMB and 80 and 150 mm for BG (Gablehouse 1984; Neumann et al. 2012). Proportional Size Distribution values of 40 - 70 for LMB and 20 - 40 for BG are considered to be indicative of a balanced population (Anderson 1980).

Limnology sampling, consisting of dissolved oxygen (DO, mg/L) and temperature (ºC) profiles, were conducted on a monthly basis from June through October. These samples were taken from a boat with a YSI model 550A meter at the surface and 1-m increments down to the bottom of the reservoir. Using an anchor, the boat was kept stationary in the deepest part of the lake while measurements were taken. These data were compared to previous data to assess potential effects of the water level drawdown on temperature and oxygen in the reservoir.

## RESULTS

### Water release evaluation

An estimated 133,164 $m^3$ of water was released from SVR in 2015, a 0.61-m reduction in reservoir surface elevation (Table 10). The in-stream residence time of water from SVC0 to SVC8 was 554 hours when 0.011 $m^3/s$ was being released from SVR from August 3-26, 2015 (Table 11). No surface flow was documented downstream in the canyon reach from LBC 10 – LBC 18 during this period. Once releases were increased to 0.027 $m^3/s$ on August 27, in-stream residence time decreased to 53 hours from SVC0 to SVC8 and 241 hours from SVC0 to LBC18 (Table 11).

An estimated 138,096 $m^3$ of water was released from SVR in 2016, a 0.62-m reduction in reservoir surface elevation (Table 10). The water release of 0.011 $m^3/s$ began on July 6, 2016. The in-stream residence time of water was 20 hours from SVC0 to SVC8 and 219 hours from SVC0 to LBC18. In 2016, since we were maintaining a minimum base flow for the study, we released around 0.013 $m^3/s$, which maintained a more consistent flow of 0.011 $m^3/s$ from SVC0 to LBC18.

Water elevation in SVR dropped 122 cm each year with approximately 61 cm attributed to water releases and 61 cm to evapotranspiration and seepage. Streamflow and timing of flow detection downstream at each sensor was dependent on the discharge amount and the start time of the flow augmentation (Figure 59 and Figure 60). When an adjustment was made at SVC0, detection time of downstream responses was dependent on the amount of flow in the stream at the time of adjustment (e.g. the more flow remaining in the stream prior to augmentation, the more immediate the response). During both years, control reaches became intermittent by mid-July.

Mean daily water temperatures at SVC0 ranged between 10 and 12°C and never exceeded 17°C downstream through the Meadow and Canyon reaches (Figure 61). Comparatively, the WFLBC control site went dry in June 2015 and mean daily temperatures at control sites averaged about 5°C warmer than the Meadow and Canyon treatment sites in 2016 during peak dry periods (mid-July through mid-August).

In 2015, mean daily DO levels in the Meadow reach ranged between 6-9 mg/L once SVR releases returned perennial flow. Prior to release treatments, DO levels averaged 2.07 mg/L (Figure 62) at SVC0. In 2015, DO concentrations downstream in the Canyon reach increased from 0.00 to 10.51 mg/L after flow augmentation returned perennial flow to this reach and then was maintained for the duration of the study. In 2016, mean daily DO concentrations remained above 6.00 mg/L in both the Meadow and Canyon reaches for the duration of the study (Figure 63). The WFLBC control site saw DO levels drop from 8.15 to 0.00 mg/L by July 1, 2015. In 2016, WFLBC control instantaneous DO decreased to 4.00 mg/L at times but averaged 9.56 mg/L throughout the study despite sensors indicating loss of flow. The BBC control sites averaged 0.43 mg/L until flow returned October 28, 2016.

The recovery rate of DO back into the stream after release from SVR was evaluated on 10/21/2015. Measured at the outlet pipe (SVC0), DO concentration was 1.65 mg/L and then steadily increased, reaching nearly 6.00 mg/L one km below the reservoir outlet and reaching a maximum of 11.00 mg/L six km downstream (Figure 64).

Prior to flow augmentation in 2015 (July), 32% of the Meadow reach, 72% of the Canyon reach, and 54% of the WFLBC control reach was wetted (Table 12). During the September 2015 survey (0.027 $m^3/s$ was being released), 100% of the Meadow and Canyon reaches were wetted while the WFLBC control was 57% wetted (Table 12). In 2015, the control sites at lower, middle, and upper Bear Creek were dry for the duration of the study. In July 2016, 100% of Meadow and Canyon reaches remained wetted throughout the study period. Wetted lengths in control reaches decreased from 100% to 29% from July to September.

In 2015, pool density in the treatment reaches was 0.72 pools/100 $m^2$ prior to the flow augmentation. Pool densities increased 28% by September, once releases reached LBC18 (Table 13). Most of the new pool creation was found between SVC2 and LBC10 (Figure 65). The July 2016 Meadow reach pool densities were 2.35 pools/100 $m^2$ and in August densities increased 35% to 3.62 pools/100 $m^2$. Throughout the 18 km of stream influenced by treatments, pool densities increased 21% from 1.76 pools/100 $m^2$ to 2.22 pools/100 $m^2$ in 2016. In comparison, control sites saw pool densities decline from 0.70 pools/100 $m^2$ to 0.01 pools/100 $m^2$.

**Reservoir evaluation**

Based on a bathymetric map and reservoir volumes calculated by DuPont et al. (2011), a 1.04-m reduction in surface elevation was measured for SVR in 2016, slightly less than the 1.07 m measured in 2015 (Figure 66; Hand et al. 2018). In both years, approximately 60% of the drawdown was attributable to the water release. Thus, approximately 0.43 m of the reduction in surface elevation was due to evaporation, seepage, etc. The reservoir surface elevation had been reduced 0.6 m by early August (Figure 66).

As the drawdown approached its maximum level (1 m), exposed banks 5 to 20 m in width (depending on the shoreline gradient) surrounded the reservoir (Figure 67). Access to docks, fishing platforms, and the boat ramp were negatively affected (Figure 67). At maximum drawdown, the lower water level caused many docks to have steep drops from shoreline to the outer dock sections, making them difficult (or impossible) to access. Similarly, fishing platforms were out of the water, making them non-functional. The boat ramp, while still accessible, was much shallower, and required people to drive farther out into the reservoir to float boats off of trailers.

Angler catch rate and harvest data was based on 69 completed trip interviews. These anglers fished a total of 118 hours, and caught a total of 69 fish. The majority of these fish were hatchery Rainbow Trout (RBT; 81%), with BG accounting for 10%, and LMB accounting for 6%. This was a catch rate of 0.6 fish/h for all species combined, similar to the catch rate of 0.7 fish/h in 2012 (August - October) with no drawdown. The RBT catch rate was 0.50 fish/h, similar to the catch rate of 0.45 fish/h in 2012 (August - October) with no drawdown. Anglers harvested 19

(28%) of the fish caught. Rainbow Trout (*n* = 17) accounted for most of the harvest, with BG (*n* = 2) and Pumpkinseed (PS) *L. gibbosus* (*n* = 2) accounting for all additional harvest. Angler opinion surveys were conducted in conjunction with the creel survey. Everyone leaving the reservoir was interviewed. Ninety-eight percent identified fishing as their primary reason for visiting SVR, compared to 58% in 2015 (Figure 68). Camping was the only other response at 2%. Anglers were also asked additional questions regarding their fishing experience. The most common targeted fish species was hatchery RBT (29%), lower than reported in 2012 and 2015 (Figure 69). Fifty-four percent of people interviewed were not targeting a particular fish species. Warm-water species comprised 17% of the targeted fish species responses for SVR, an increase from previous surveys (Figure 69). Anglers rated their fishing experience as excellent/good during drawdown years at 63% (2015) and 83% (2016), both higher than the 58% (2012) when there was no drawdown (Figure 70). The most common reasons for a positive rating were “good location/amenities” (35%), and “good fishing” (25%). Additionally, anglers rated their fishing experience as fair/poor at 17% (2016) and 36% (2015) during drawdown years, both lower than the 42% (2012) when there was no drawdown (Figure 70). The most common reasons for a negative rating were related to poor fishing (12%). This represents a continued drop from 30% in 2012 and 19% in 2015.

A total of 273 fish were sampled through the electrofishing survey, including LMB (*n* = 66), BG (*n* = 79), PS (*n* = 113), Black Crappie *Pomoxis nigromaculatus* (*n* = 6), and Black Bullhead *Ameiurus melas* (*n* = 9; Table 14). The CPUE for all fish sampled was 411 fish/h (±116; Figure 72). This was the second lowest CPUE for surveys conducted from 1997 to 2016. The mean WPUE (90% C.I.) of 35.3 kg/h (± 14.6) was significantly lower than the 86.2 kg/h (±18.5) in 2012, and was the second lowest since 2001 (Figure 73).

Largemouth Bass CPUE (99 fish/h, ±26) was among the lowest seen for sampling conducted since 1993 (Figure 74). The WPUE for LMB was 9.8 kg/h (±1.7), significantly lower than 2015 (26.1; ±3.2) and the lowest of any survey conducted since 2001 (Figure 75). Largemouth Bass sampled ranged from 85 to 357 mm in length, with an average length of 191 mm (±13; Figure 76). Largemouth Bass PSD was 8 similar to 2015 (PSD = 6), the lowest value of any survey from 1997 to 2016 (Figure 77).

Bluegill CPUE (119 fish/h, ±37) was the lowest for any survey from 1997 to 2016 (Figure 74). The WPUE for BG was 5.8 kg/h (±2.9), significantly lower than 2012 (44.8; ±14.3) and the lowest of any survey conducted since 2001 (Figure 75). Bluegill sampled ranged from 40 to 222 mm in length, with an average length of 110 mm (±10; Figure 78). Bluegill PSD was 51, which was the fifth consecutive increase since the low of 15 in 2006 (Figure 77).

Pumpkinseed CPUE (170 fish/h, ±82) was the highest for any survey from 1997 to 2016 (Figure 74). The WPUE for PS was 10.7 kg/h (±6.7), the highest for any survey (Figure 75). Pumpkinseed sampled ranged from 46 to 170 mm in length, with an average length of 129 mm (±6; Figure 79). Pumpkinseed PSD was 12.

Limnology sampling was conducted in August 2015 and 2016 to compare with data collected in years prior to drawdown (Table 15; Hand et al. 2016; Hand et al. 2018). We compared August data as this is when the drawdown is mostly complete and is when the most stressful conditions for fish will usually occur. Sampling indicated that the thermocline occurred around 3 - 4 m of depth in all years (Table 15). Dissolved oxygen (DO) levels remained above 5.0 mg/L in the epilimnion before and during the drawdown. Water temperature was above 20 °C in the upper portion of the epilimnion during all years, but were lower in years with drawdown. This is likely due to environmental factors, not the drawdown, as water is released from the bottom of the reservoir in the hypolimnion.

## DISCUSSION

### Water release evaluation

In-stream residence time of water released from SVR was at least 73 times faster when released in a channel that had perennial rather than intermittent flow. The difference in residence time can likely be explained by the dewatering of the hyporheic zone in 2015 before flow augmentation commenced. Before water releases could impact stream surface water (temperature, dissolved oxygen, wetted length, pool density) the hyporheic zone needed to be recharged. In-stream responses to flow augmentation may also be influenced by evapotranspiration, and the amount of time it takes to fill pools (Brooks and Treasure 2014). By maintaining a charged hyporheic zone in 2016, streamflow over 10 km downstream responded to increased water releases in under two days. A flow augmentation study in a nearby stream (Big Meadow Creek) showed similar responses to streamflow under intermittent conditions (Brooks and Treasure 2014). After Big Meadow Creek went completely dry, it took 24 hours after water was released to detect surface flow increases 2 km downstream and 48 hours to be detected 6 km downstream (Brooks and Treasure 2014). To maximize the response to flow augmentation, releases should begin while perennial streamflow remains in SVC.

Releases of 0.007 - 0.014 $m^3/s$ from SVR maintained perennial flow 18 km downstream throughout the peak dry months. Perennial flows were maintained throughout the study area by ensuring that flows of at least 0.006 $m^3/s$ occurred at the end of the Meadow reach 8 km downstream (SVC8). In nearby Big Meadow Creek, reservoir water releases of 0.010 $m^3/s$ maintained perennial flows of at least 0.005 $m^3/s$ 10 km downstream (Brooks and Treasure 2014). Loss of surface flow in a stream can be explained by evaporation and evapotranspiration as influenced by native vegetation composition, land use practices, percolation, and diversions for water rights uses (Beechie et al. 2010, Brooks and Treasure 2014). Adjustments in the amount of water released from SVR to maintain perennial flow 18 km downstream will likely depend on changes in instream losses over time.

Spring Valley Reservoir has limited capacity and multiple uses, thereby limiting the maximum amount of water available for release without impacting recreation. Up to 0.028 $m^3/s$ was released in 2015, but sustainable release strategies will need to ensure that drawdown at the reservoir does not degrade recreation. The strategy used in both years (in addition to evaporation and seepage), led to a 122 cm drawdown at SVR and managers will need to consider if this is acceptable or if alternative strategies should be considered.

During the study, temperatures recorded downstream of SVR never exceeded 17°C and were consistently cooler than in nearby control streams, owing, at least partially, to the cold water (10-12°C) being released from the reservoir. Similar results were witnessed in Big Meadow Creek, where water temperatures were typically cooler (up to 8 °C) in flow-augmented reaches than control sites during peak dry periods. (Brooks and Treasure 2014).

Dissolved oxygen concentration increased from about 2 mg/L at the reservoir outlet to nearly 6 mg/L 1 km downstream of SVR. Dissolved oxygen concentrations were maintained at or above 6 mg/L, the Idaho standard for cold water biota (Brooks and Treasure 2014), throughout the entire 18 km study reach once flow augmentation returned perennial flow to the stream. The ideal range of DO for steelhead ranges between 6-9 mg/L (Carter 2005, USEPA 1986).

A lack of water retention and the resulting lack of late summer rearing habitat is one of the primary limiting factors to steelhead populations in the lower Potlatch River drainage (Banks and Bowersox 2015, Uthe et al. 2017). Stream flow decreased 94.6% in the lower Potlatch River basin from early summer (June/July) to base flows (Banks and Bowersox 2015). Further, life cycle modeling in the Big Bear basin suggests increasing the amount of available access to reaches with adequate water quality and habitat via flow augmentation could increase steelhead smolt production by 2,500 smolts annually (Uthe et al. 2017). These results suggest that flow augmentation timed prior to the stream drying up in summer can provide this critical habitat across the entire 18-km study area. Prior to flow augmentation restoring 100% wetted length by September 2015, the Meadow (29% wetted) and Canyon reaches (72% wetted) lacked wetted connectivity. In 2016, 100% wetted length was maintained throughout summer by beginning releases two months earlier, at the beginning of June, once base flows reached 0.006 $m^3/s$. By watering the stream while there is still lateral storage in the floodplains of the Meadow reach, this elevated water table continued allowing storage water inputs into the dry season.

Rearing habitat is severely limited in the Potlatch Basin by tributary blockages and low flow, and movement of rearing steelhead within the watershed is difficult by mid- to late-summer (Uthe et al. 2017). Hartson and Kennedy (2015) found growth, survival and movement of juvenile steelhead varied in response to a complex combination of intraspecific competition and reduced flows in Lapwai Creek, another large tributary of the lower Clearwater River with very similar characteristics to the Potlatch River. In 2015, pool densities increased by 117% and 86% in the Meadow and Canyon reaches respectively (Table 13), once these sections were wetted via flow augmentation. Concomitantly, pool densities in control reaches decreased by 25% or more. In 2016, pool density increased from 1.8 pools/100 $m^2$ to 2.2 pools/100 $m^2$ in treatment reaches while pool density decreased from 0.61 to 0.17 pools/100 $m^2$ in control reaches. The increase in pools within flow augmented reaches may provide critical rearing habitat for juvenile steelhead and reduce intraspecific competition within populations. Total pools counted were 33% and 79% higher in the Meadow and Canyon reaches with the 2016 flow augmentation that kept perennial flow, suggesting this release strategy can provide more habitat throughout peak summer low water periods (Figure 65).

**Reservoir evaluation**

The primary concerns with a drawdown of SVR are impacts to recreational use of the reservoir (access, effort, catch rates, etc.) and to the fishery. Reduced access would be in the form of more difficulty for anglers to access docks/fishing platforms, boat ramps, and shore fishing. With a 1-m decrease in surface elevation and exposed banks up to 20 m in width, accessibility to these amenities was decreased (Figure 67). The high angle on the walkways to the docks, and lack of water near shore could be causing access problems for many anglers. Additionally, there is concern that some people could stop going to the reservoir knowing the water levels are low. As we would not have the opportunity to interview these people, our surveys may have some inherent bias since the people we are interviewing could be those that are less likely to have an issue with water levels. Although our lowland reservoirs often have reduced angler effort in late summer months, at SVR 17.5% of effort in 2012 occurred during August and September (Hand et al. 2016a). If the drawdown becomes an issue for anglers, this could impact a portion of the effort in this reservoir.

Based on our angler surveys, catch rates for warm-water fish and RBT were similar to pre-drawdown surveys, indicating no negative effect on angling. However, our results should be interpreted with caution due to low sample size. As such, we were unable to calculate an effort estimate that would comparable to previous creel surveys. If this project is continued, we should conduct a full creel survey to allow for a better comparison of catch rates and effort before and during drawdowns. Angler opinion surveys indicated no negative responses to the drawdown during 2016, and only a few negative responses in 2015 (most of which were concerned about the county taking water from the reservoir for dust abatement, not impacts on fishing). As with 2015, the public may have assumed that the low water was due to the natural annual drop in water level caused by evaporation and seepage. While this suggests that the drawdown is not directly affecting angler satisfaction, other responses could have just been more important. In fact, if the drawdown is affecting fishing and/or amenities, anglers could be responding with a negative angler satisfaction response about those categories instead of directly attributing it to the drawdown. In the future, we need to ask more specific questions regarding angler's opinions of the drawdown and its potential effects on their experience.

The fish population survey indicated that although PSD of LMB was similar to 2015, it has been much lower during drawdown years than pre-drawdown (Figure 77). In contrast, BG PSD has increased steadily since 2006 (Figure 22). This could indicate that the drawdown is negatively impacting the fishery. However, we would expect the drawdown to have a higher impact on young fish through predation. Additionally, this impact would not be realized for several years until those fish fully recruited to our sampling gear. With CPUE higher than recent pre-drawdown surveys, the low LMB PSD appears to be the result of fewer fish >300 mm captured as opposed to a general loss of the population (Figure 76). This is a noticeable change from the 2012 survey, and is likely the result of changes in mortality (natural and/or angler) or variable recruitment. Some of this could be due to increased angler harvest, but a more robust angler survey would be needed to determine if this is the case. Regardless, variable recruitment is likely a driving force in the decline in PSD. The lack of LMB >300 mm suggests that there was a period of poor recruitment that led to few fish available to replace the mortality loses. However, with numerous fish in the 150-300 mm range, we would expect to see increases in the number of fish >300 mm (and thus improvements in PSD) over the next few years. Again, we caution that the effects of drawdown may take several more years to become apparent. Thus, we recommend conducting additional evaluations if the drawdown program continues.

Drawdowns are often used intentionally to manage fish populations. They can stimulate fish productivity by reestablishing conditions similar to when a reservoir was first filled (Miranda and Muncy 1987; Cooke et al. 2005). Other potential effects are increased predation on stunted prey populations, reduced predation on eggs by Centrarchids, and reduced competition for resources for young-of-year Largemouth Bass (Heman et al. 1969; Miranda et al. 1984). The result can be improved sport fisheries through increased biomass and sizes of game fish, and a reduction in abundance of stunted BG, crappie, or other planktivores. These effects of water level drawdown are likely contributing to the quality warm-water fishery found in Mann Lake (Hand et al. 2016). If the drawdowns at SVR show similar positive effects, they may actually contribute to improving the fishery in the future.

The trends in DO and temperature were similar to previous samples, indicating that the drawdown conducted in 2015 - 2016 did not have a negative impact on these limnological variables. As such, we do not have any concerns that the drawdown at current levels will negatively affect these variables in a way that could impact the fisheries in SVR.

### Future direction

This project indicated that a sustained 0.014 $m^3/s$ release from SVR during peak summer low water periods can provide the improvements in temperature, DO, stream flow, and habitat availability to enhance juvenile steelhead survival in Spring Valley and Little Bear Creek in the 18-km reach below SVR. Water releases timed before the dewatering of the hyporheic zone resulted in perennial flows throughout the summer dry periods, decreased water temperature, increased DO concentration, and rapid increases in pool density and wetted length of the entire study area. Additionally, there were no apparent negative impacts on the fish populations or fishery.

If this drawdown becomes a long-term project, being proactive to ensure continued access and use of the reservoir and facilities will be necessary. Increasing storage capacity of the reservoir would help maintain recreational use of SVR while providing a water source to sustain perennial flow in Spring Valley and Little Bear Creek through annual summer dry periods. Potential modifications include raising the height of the dam to increase storage, which will undoubtedly require improvements to recreational access and infrastructure. This would include projects such as reducing the angle of the walkways to the docks, extending the lengths of the docks so they extend further into the water, adding stepped concrete fishing platforms, and extending the boat ramp. Additionally, it is also possible that if the length of low water period (mid-June – mid-September) increases over time due to environmental effects of climate change, the reservoir may need additional storage to avoid reducing outflow too early in the fall. These, and other potential alternatives, will require further research to develop the best option(s) based on available funding. While flow augmentation will require modifications to lake access and amenities so as not to degrade the recreational value of Spring Valley Reservoir, we believe that it is a viable management tool to increase critical summer rearing habitat and connectivity for juvenile steelhead.

## MANAGEMENT RECOMMENDATIONS

1. Flow supplementation is a plausible technique for reducing density-dependent effects and increasing steelhead population productivity in Spring Valley and Little Bear Creek. We found managing the release from the reservoir at 0.014 $m^3/s$ flow for 3 months provided a perennial flow in an otherwise dry system.

2. Begin flow release from the reservoir before the stream dries out when base flows reach 0.006 $m^3/s$ at SVC8. Downstream response to flow augmentation should continue to be monitored and if necessary, releases increased, to ensure temperature, DO, and flow are maintained below state cold water biota criteria.

3. Evaluate options to increase storage capacity of the reservoir, and to maintain recreational access during drawdowns.

4. We recommend continuation of drawdown program only if improvements to storage capacity and access of the reservoir are addressed.

Table 9. Questions asked during angler opinion surveys at Spring Valley Reservoir, Idaho, in 2015 and 2016.

1. Do you have a current hunting or fishing license?
2. What was your primary reason for visiting this lake today?
3. How would you rate your fishing experience today?
4. What species are you targeting today?
5. Give your top reason that led to the rating you gave for your fishing experience.

Table 10. Estimated volume of water released from Spring Valley Reservoir, Idaho and the associated drawdown elevation during a flow augmentation study in 2015 and 2016.

| 2015 | Flow ($m^3$/s) | $m^3$ released | Reservoir drawdown (cm) |
|---|---|---|---|
| 8/3/2015 - 8/26/2015 | 0.011 | 23,470 | 10.65 |
| 8/27/2015 - 9/25/2015 | 0.028 | 71,430 | 32.61 |
| 9/26/2015 - 10/1/2015 | 0.011 | 5,867 | 2.74 |
| 10/2/2015 - 10/8/2015 | 0.027 | 16,667 | 7.62 |
| 10/9/2015 - 10/21/2015 | 0.014 | 15,604 | 7.01 |
| | Avg: 0.018 | Total: 133,038 | Total: 60.63 |

| 2016 | Flow ($m^3$/s) | $m^3$ released | Reservoir drawdown (cm) |
|---|---|---|---|
| 6/6/2016 - 6/12/16 | 0.015 | 8,905 | 3.96 |
| 6/13/2016 - 6/27/16 | 0.007 | 10,178 | 4.57 |
| 6/28/2016 - 9/26/16 | 0.013 | 104,645 | 47.85 |
| 9/27/2016 - 10/10/16 | 0.011 | 13,018 | 6.1 |
| | Avg: 0.012 | Total: 136,746 | Total: 62.48 |

Table 11. In-stream residence time (hours) of water released from Spring Valley Reservoir, Idaho to monitoring sites 0-18 km downstream in 2015 and 2016.

| | Date | Aug 3, 2015 | Aug 26, 2015 | July 6, 2016 |
|---|---|---|---|---|
| | Mean Daily Discharge ($m^3/s$) | 0.014 | 0.028 | 0.013 |
| | Site | Residence Time | Residence Time | Residence Time |
| Meadow Reach | SVC0 | 0 | 0 | 0 |
| | SVC2 | 38 | 4 | 4 |
| | SVC4 | 74 | 11 | 14 |
| | SVC6 | 218 | 16 | 16 |
| | SVC8 | 554 | 53 | 20 |
| Canyon Reach | LBC10 | NA | 101 | 44 |
| | LBC12 | NA | 193 | NA |
| | LBC14 | NA | 212 | NA |
| | LBC16 | NA | 241 | 118 |
| | LBC18 | NA | NA | 216 |

Table 12. Wetted stream length (%) in the Meadow and Canyon treatment reaches and control sites. *Due to loss of flow in 2015, there was only 1 control site (WFLBC) while all 4 control sites had measurable wetted length in 2016.

| | Date/Reservoir Discharge ($m^3/s$) | % Wetted Meadow Reach | Canyon Reach | Control Site(s) |
|---|---|---|---|---|
| July 2015 | No Discharge | 29% | 72% | *54% |
| Sep 2015 | 0.028 | 100% | 100% | *57% |
| July 2016 | 0.013 | 100% | 100% | 100% |
| Aug 2016 | 0.013 | 100% | 100% | 29% |

Table 13. Pool density (pools/100 m²) during a flow augmentation study in 2015 and 2016 at each 200 m survey site downstream from Spring Valley Reservoir, Idaho, in comparison to control sites.

| | | Pool density | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Controls | | | | Meadow | | | | Canyon | | | |
| Date | Discharge (m³/s) | Lower BBC | Middle BBC | Upper BBC | WF LBC | SVC 2 | SVC 4 | SVC6 | SVC 8 | LBC10 | LBC 12 | LBC 14 | LBC 16 |
| 15-Jul | 0.000 | NA | NA | NA | 0.9 | 0.0 | 2.6 | 0.0 | 0.2 | 0.92 | 0.34 | 0.96 | 0.76 |
| 15-Sep | 0.027 | NA | NA | NA | 0.7 | 1.6 | 1.9 | 1.9 | 1.2 | 2.35 | 0.55 | 0.36 | 1.21 |
| 16-Jul | 0.013 | 0.1 | 0.5 | 0.7 | 2.3 | 3.2 | 2.1 | 2.1 | 2.0 | 2.19 | 0.59 | 0.59 | 1.29 |
| 16-Aug | 0.013 | 0.0 | 0.0 | 0.5 | 0.0 | 4.3 | 4.5 | 4.0 | 1.7 | 1.7 | 0.52 | 0.35 | 0.71 |

Table 14. Number of fish collected in each 10-minute sample, and catch-per-unit-effort (CPUE; fish/h) with 90% confidence intervals (CI) for an electrofishing survey of Spring Valley Reservoir, Idaho, in 2016.

| | Count of fish collected | | | | | |
|---|---|---|---|---|---|---|
| Species | EF Sample 1 | EF Sample 2 | EF Sample 3 | EF Sample 4 | CPUE (fish/h) | 90% CI |
| Largemouth Bass | 16 | 12 | 24 | 14 | 66 | 26 |
| Bluegill | 27 | 11 | 25 | 16 | 79 | 37 |
| Pumpkinseed | 47 | 35 | 23 | 8 | 113 | 82 |
| Black Bullhead | 1 | 2 | 6 | 0 | 9 | 13 |
| Black Crappie | 0 | 0 | 5 | 1 | 6 | 12 |
| Total | 91 | 60 | 83 | 39 | 273 | 116 |

Table 15. Dissolved oxygen (DO, mg/L) and temperature (ºC) profiles for Spring Valley Reservoir, Idaho, in August, 2009 - 2016.

| | 2009 | | 2012 | | 2015 | | 2016 | |
|---|---|---|---|---|---|---|---|---|
| Depth | D.O | Temp | D.O | Temp | D.O | Temp | D.O | Temp |
| 0m | 12.5 | 23.0 | 9.70 | 25.2 | 6.49 | 21.2 | 8.1 | 20.9 |
| 1m | 12.7 | 22.9 | 10.00 | 24.7 | 6.55 | 20.5 | 7.8 | 20.6 |
| 2m | 12.5 | 22.1 | 10.30 | 24.1 | 6.72 | 20.3 | 6.8 | 20.4 |
| 3m | 5.7 | 19.8 | 12.98 | 22.8 | 1.55 | 19.9 | 5.3 | 20.0 |
| 4m | 0.6 | 16.8 | 3.30 | 19.4 | 0.07 | 15.9 | 0.9 | 19.0 |
| 5m | 0.3 | 12.1 | 0.04 | 13.8 | 0.04 | 15.4 | 0.1 | 15.2 |
| 6m | 0.3 | 9.9 | 0.02 | 10.8 | --- | --- | 0.1 | 12.8 |
| 7m | 0.3 | 8.7 | 0.01 | 9.4 | --- | --- | 0.1 | 11.4 |
| 8m | 0.3 | 8.3 | 0.01 | 8.8 | --- | --- | --- | --- |

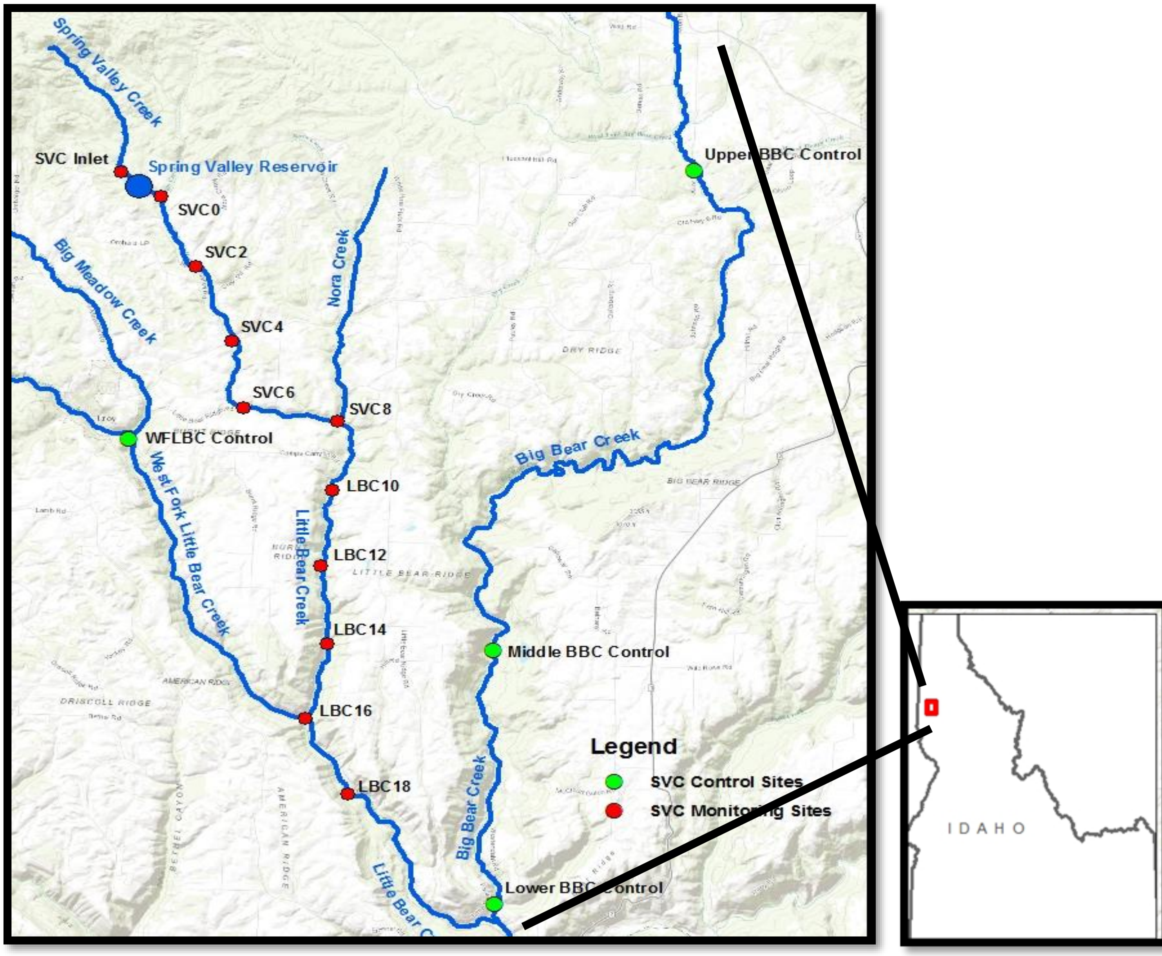


Figure 56. Map of the 2015-16 Spring Valley Reservoir flow augmentation study area within the Big Bear Creek watershed, Idaho. Green and red dots show locations of monitoring sites in treatment and control reaches, respectively.

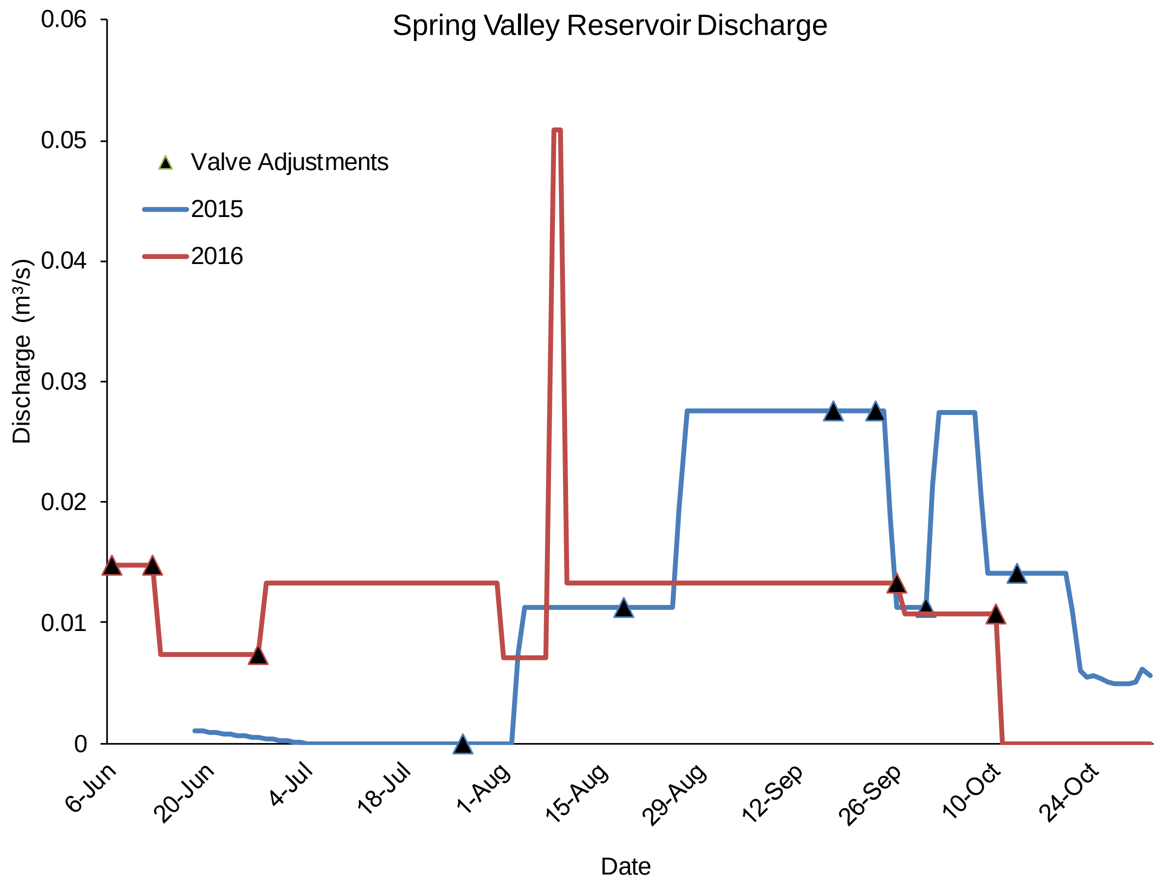


Figure 57. Discharge from Spring Valley Reservoir, Idaho, during a flow augmentation study in 2015 and 2016.


Figure 58. Locations of starting points for electrofishing survey sub-samples on Spring Valley Reservoir, Idaho, in 2016.

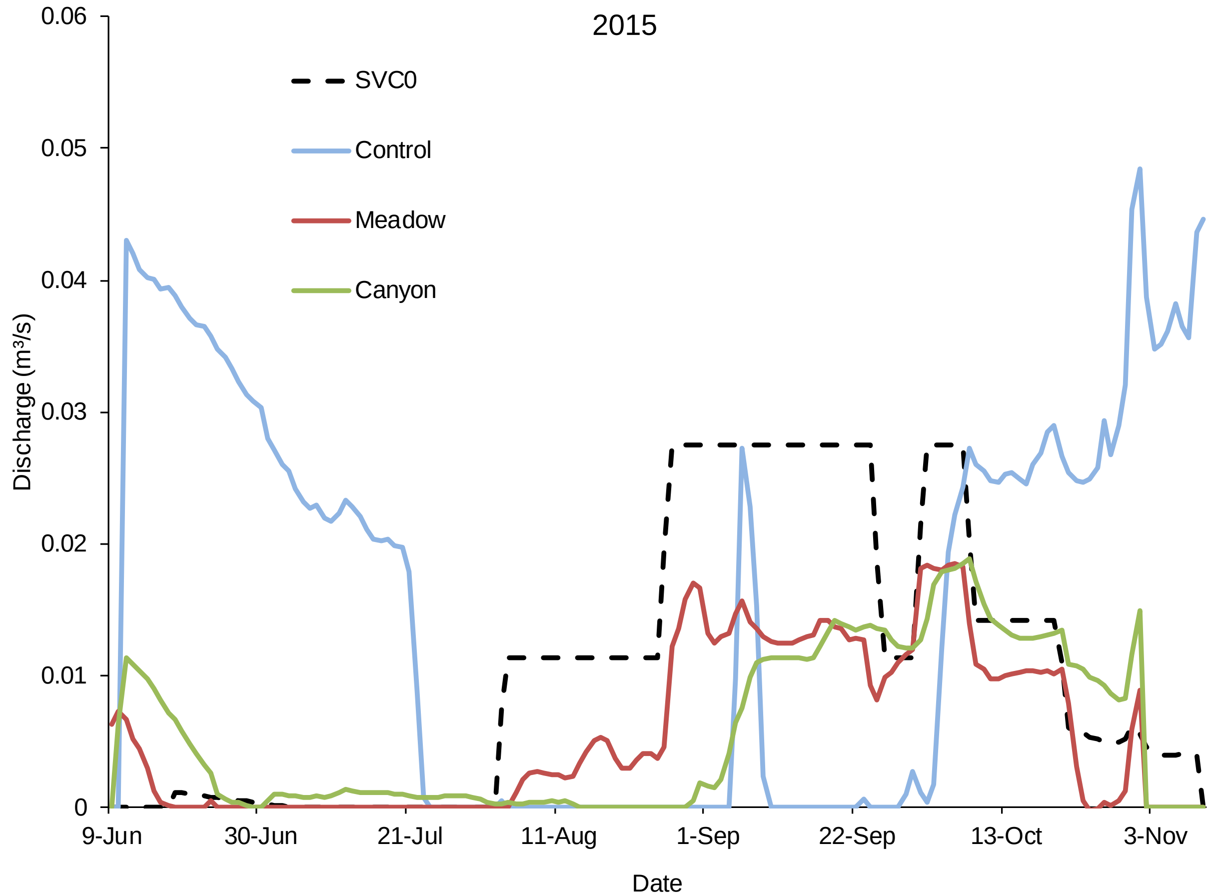


Figure 59. The 2015 discharge profile from Spring Valley Reservoir, Idaho, in relationship to downstream flows recorded at sample sites in the Meadow (SVC 2-8), Canyon (LBC 10-16) and control site in West Fork Little Bear Creek. Water released at the SVR outlet pipe (SVC0), is represented by the dashed black line.

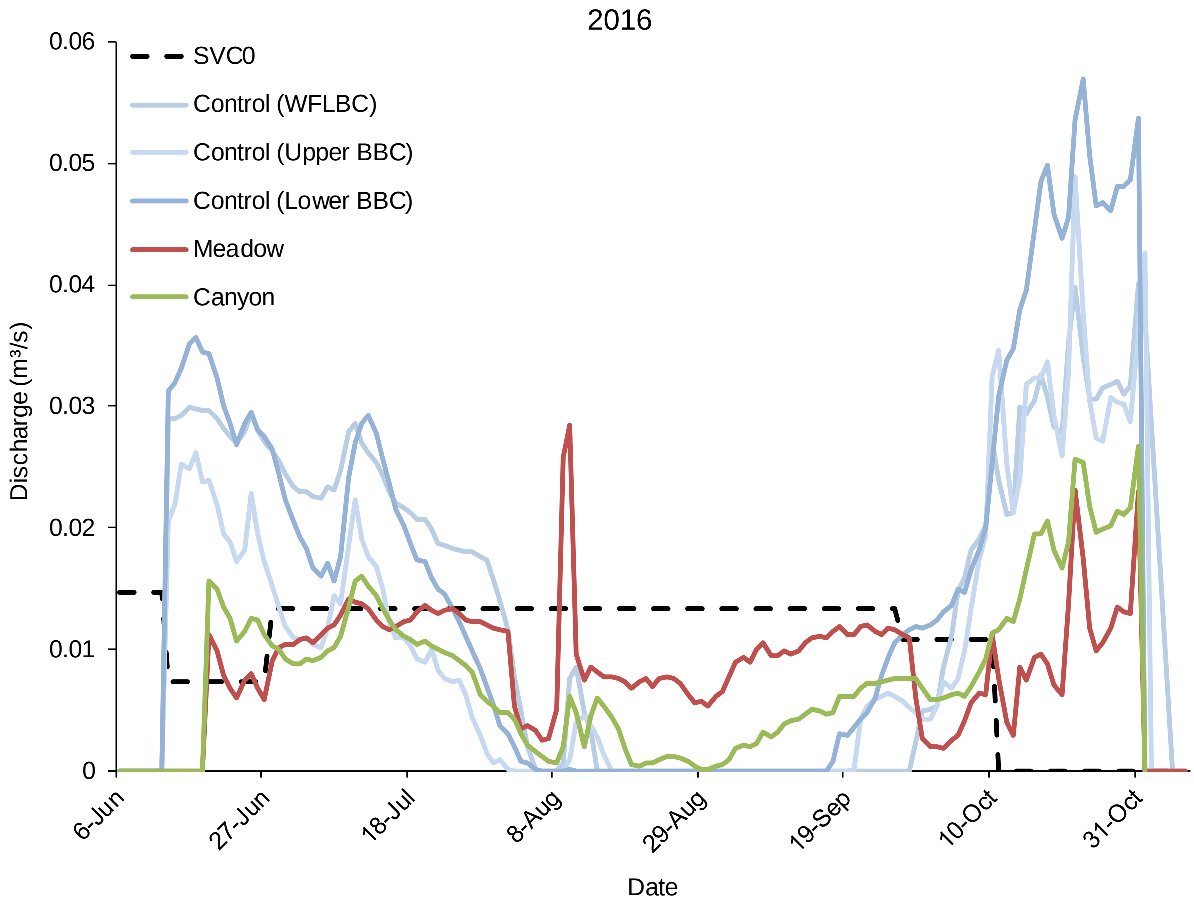


Figure 60. The 2016 discharge profile from Spring Valley Reservoir, Idaho, in relationship to downstream flows recorded at sample sites in the Meadow (SVC 2-8), Canyon (LBC 10-18) and control sites in West Fork Little Bear Creek (WFLBC) and Big Bear Creek (Upper and Lower BBC). Water released at the SVR outlet pipe (SVC0), is represented by the dashed black line.

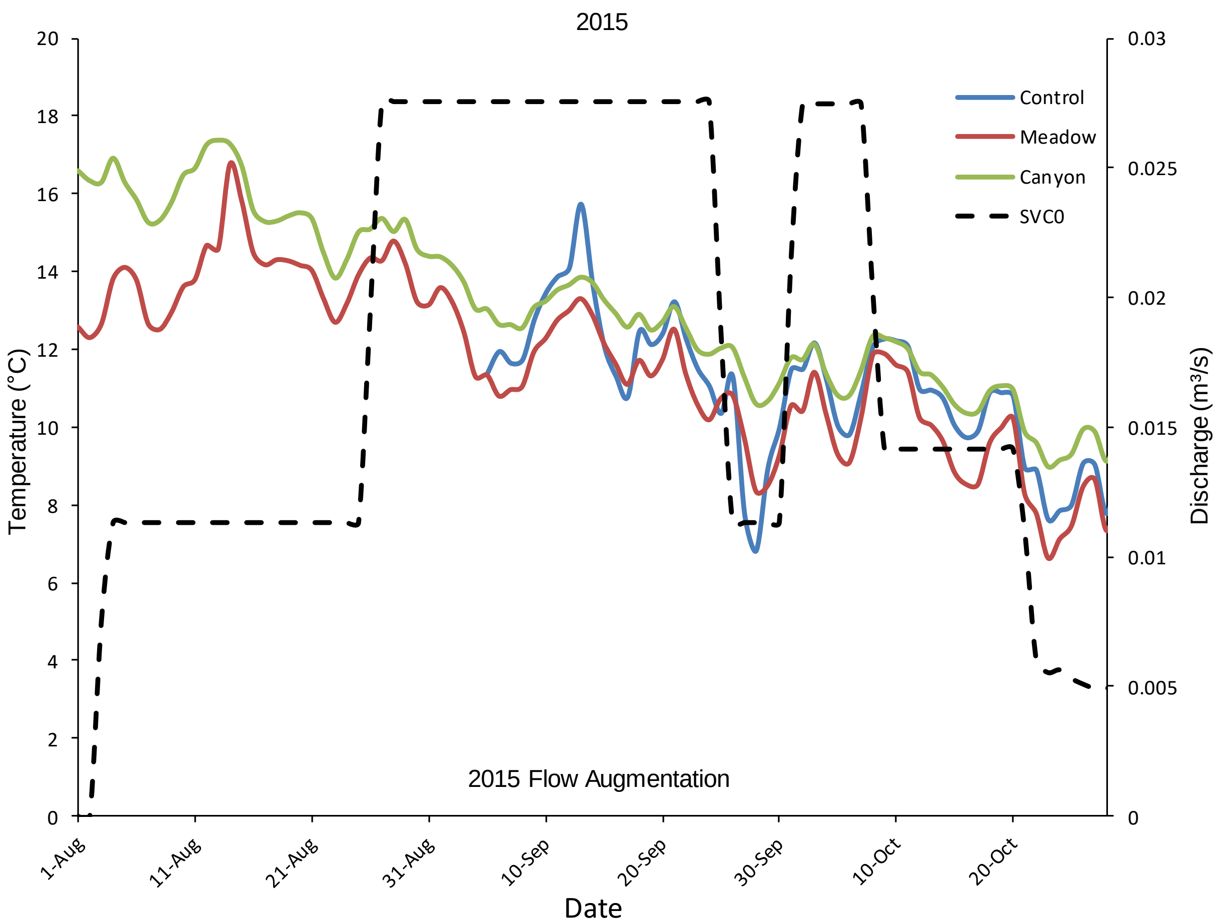


Figure 61. Mean daily stream temperatures collected during flow augmentation study in 2015 downstream of Spring Valley Reservoir, Idaho, sample sites in the Meadow reach (SVC 2-8) and Canyon reach (LBC 10-18), and the control site in WFLBC. The control site was dry until September 5, 2015. Water released at the SVR outlet pipe (SVC0), is represented by the dashed black line on the Z axis.

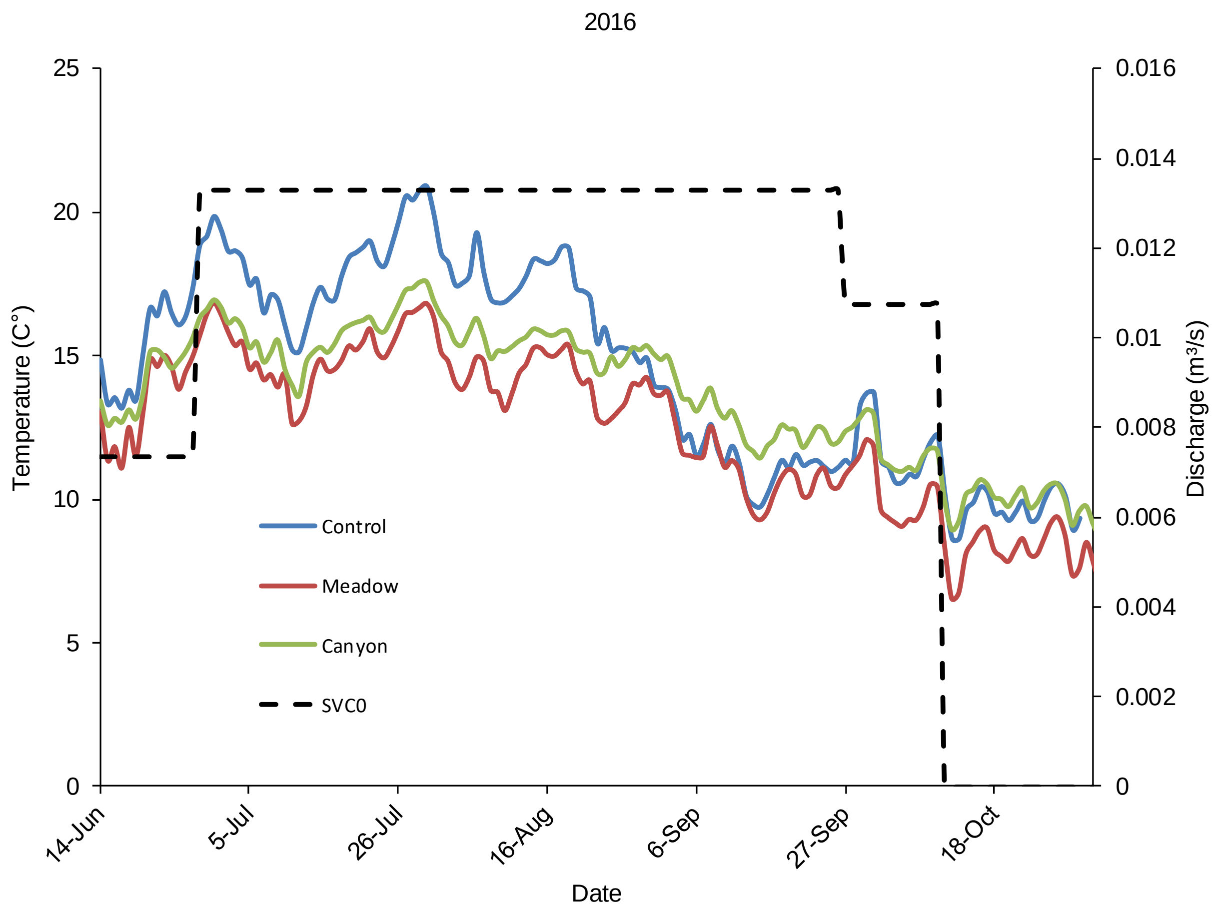


Figure 62. Mean daily stream temperatures collected during flow augmentation study in 2016 (bottom panel) downstream of Spring Valley Reservoir, Idaho, at the outlet pipe (SVC0), sample sites in the Meadow reach (SVC 2-8) and Canyon reach (LBC 10-18), and at control sites. Water released at the SVR outlet pipe (SVC0), is represented by the dashed black line on the Z axis.

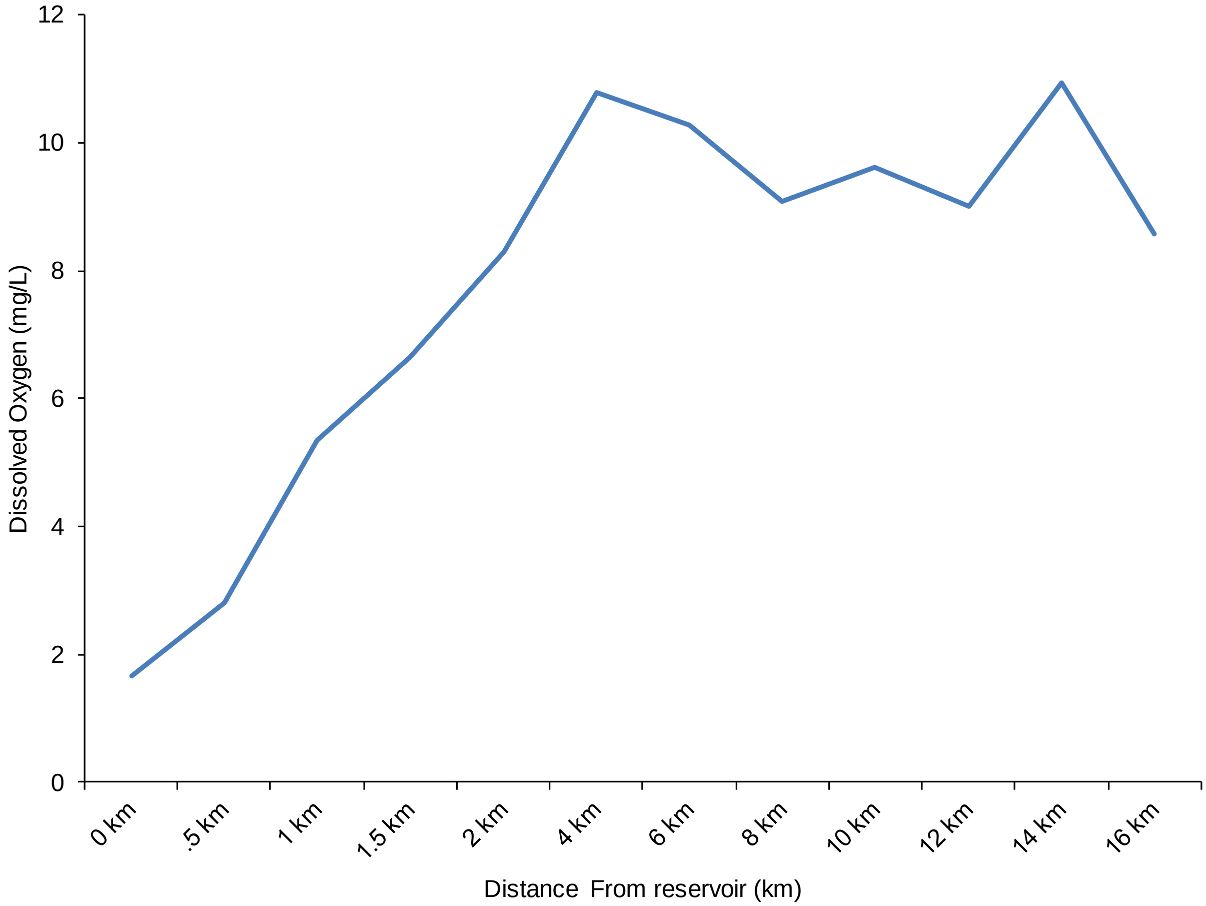


Figure 63. Recovery of Dissolved Oxygen concentrations downstream of Spring Valley Reservoir on 10/21/2015 after being released in a flow augmentation study.

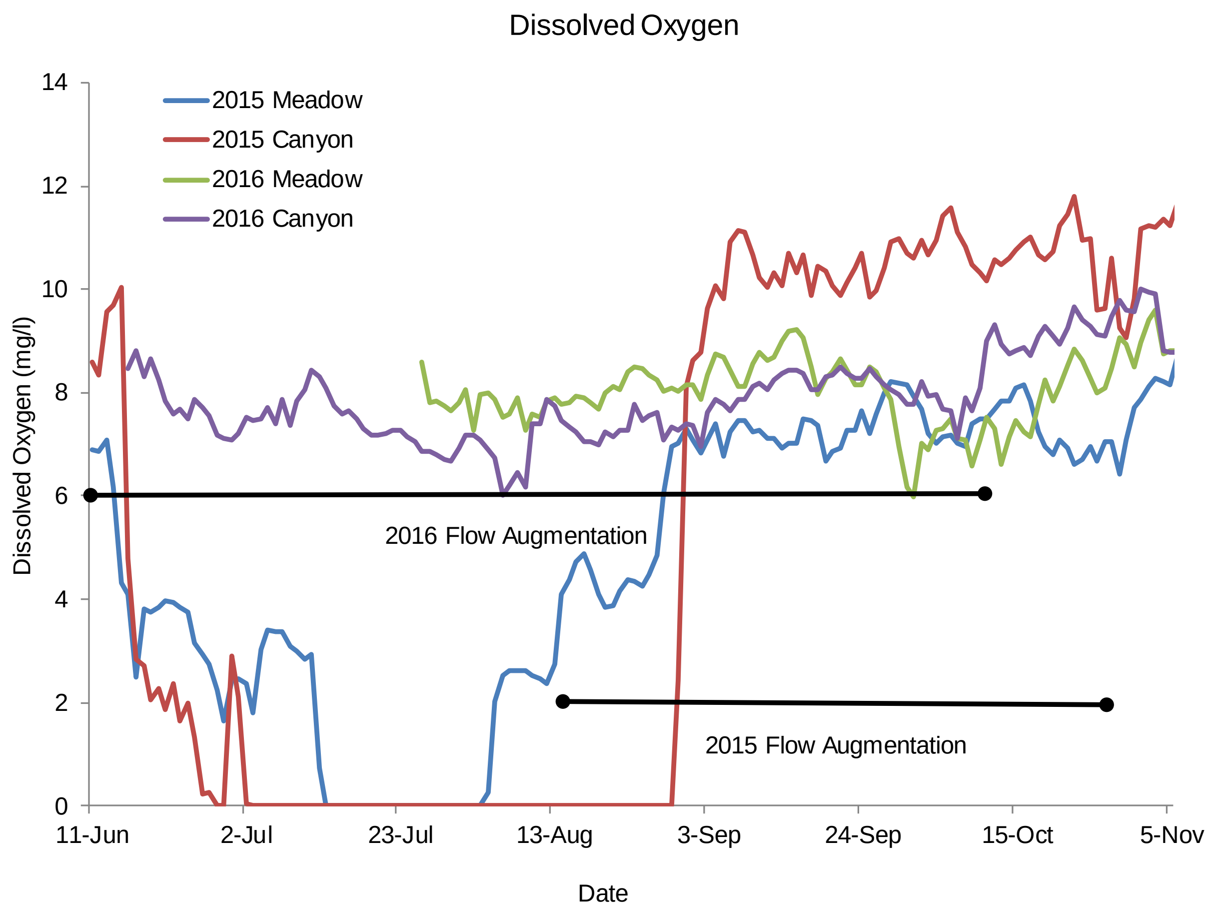


Figure 64. Mean daily dissolved oxygen levels collected during a flow augmentation study downstream of Spring Valley Reservoir, Idaho, in the Meadow (SVC 0-8) and Canyon (LBC 10-18) reaches in 2015 and 2016. In 2016 the DO sensors were not operational in the Meadow until July 26.

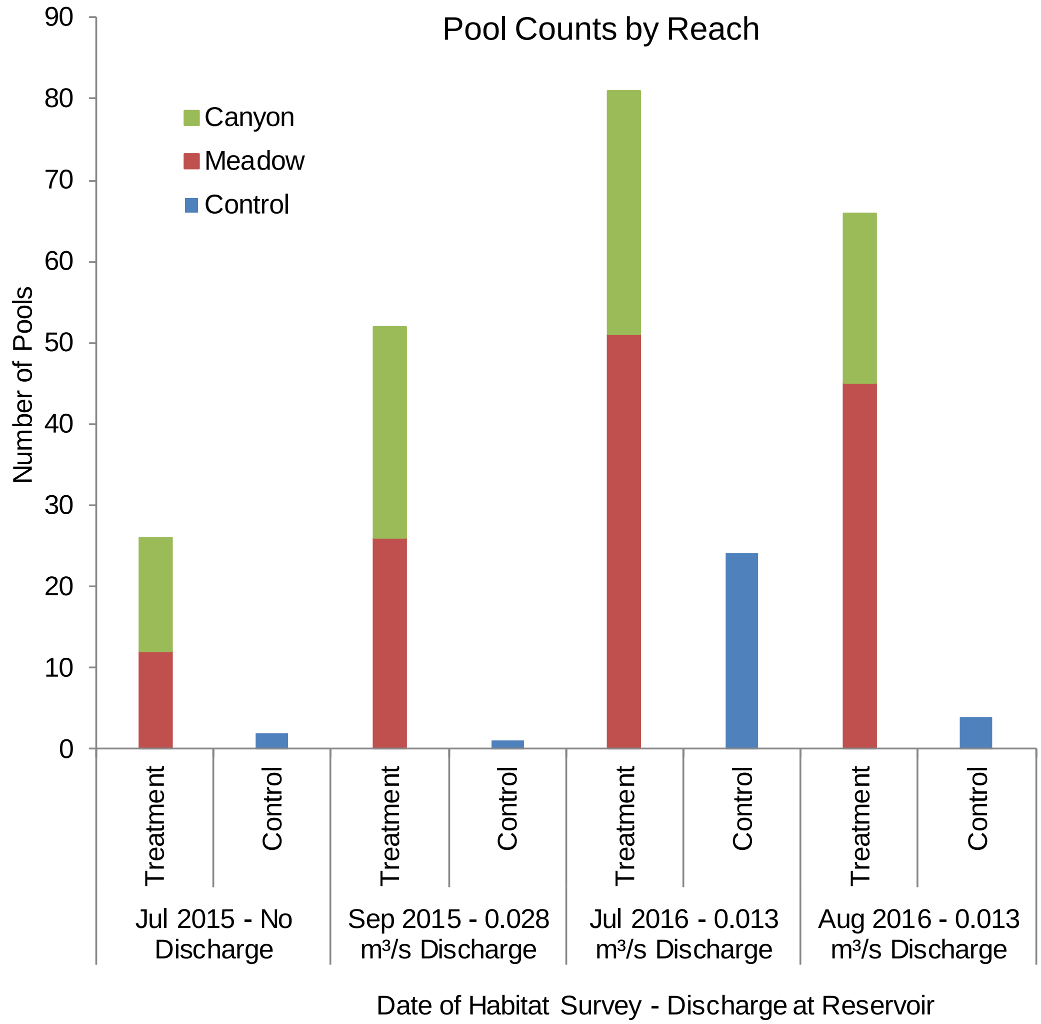


Figure 65. Comparison of changes in pool quantities in treatment vs. control reaches in response to flow releases in 2015 and 2016.

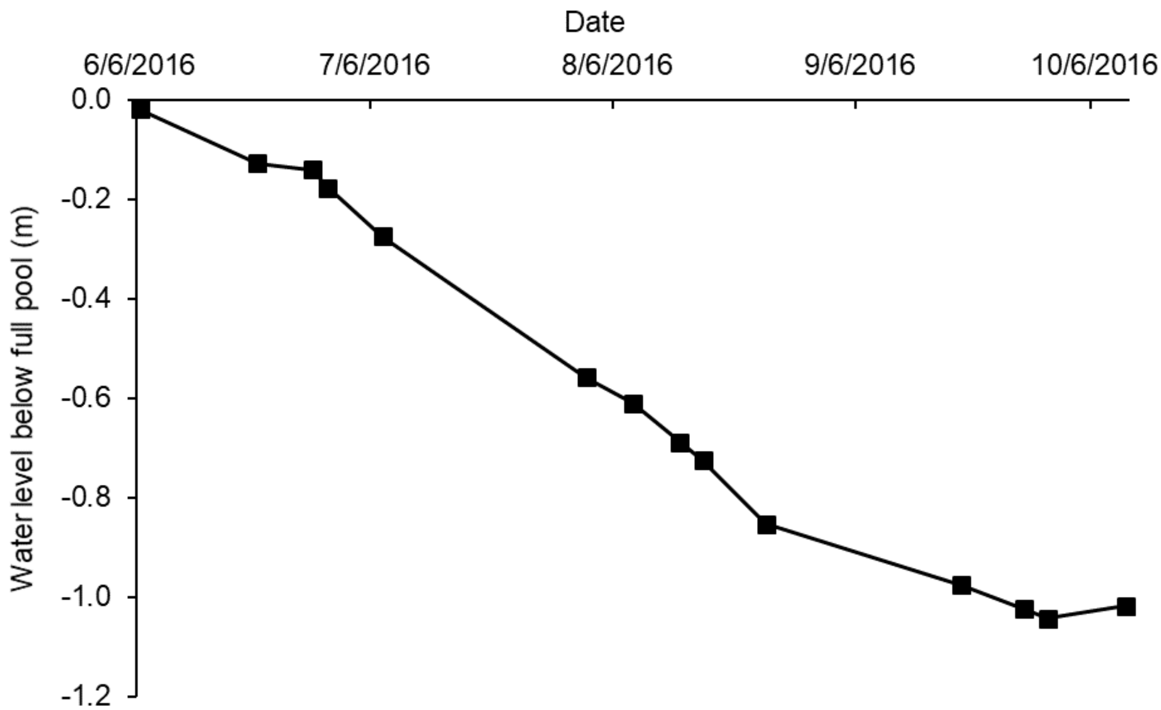


Figure 66. Water level of Spring Valley Reservoir, Idaho, during the drawdown conducted from June 6 to October 10, 2016.

Figure 67. Photos taken September 28, 2016, showing the maximum effects of the 1.0 m reduction in surface elevation on Spring Valley Reservoir, Idaho, after water release was conducted.

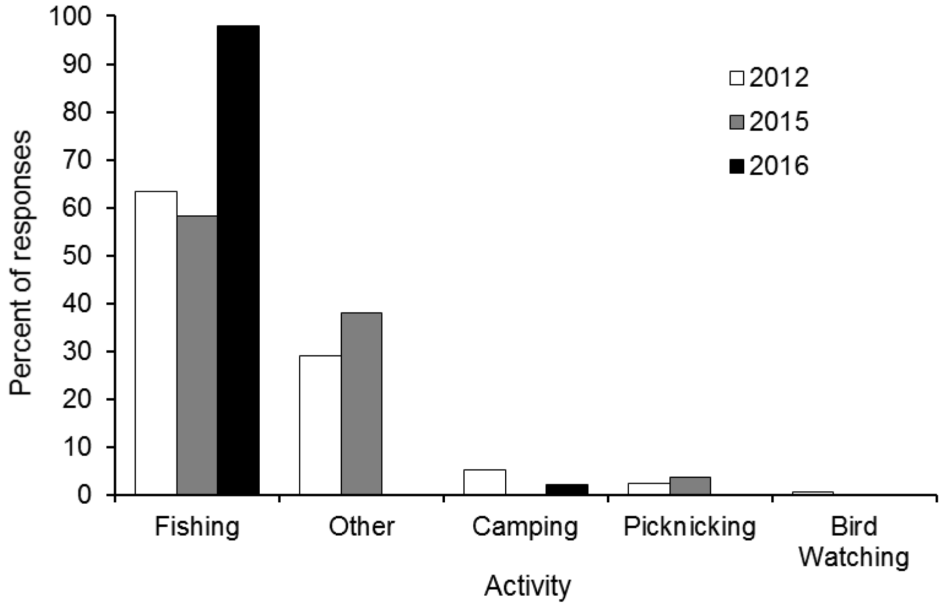


Figure 68. Comparison of angler responses collected during creel surveys regarding the primary reason for visiting Spring Valley Reservoir, Idaho, in 2012, 2015 and 2016.

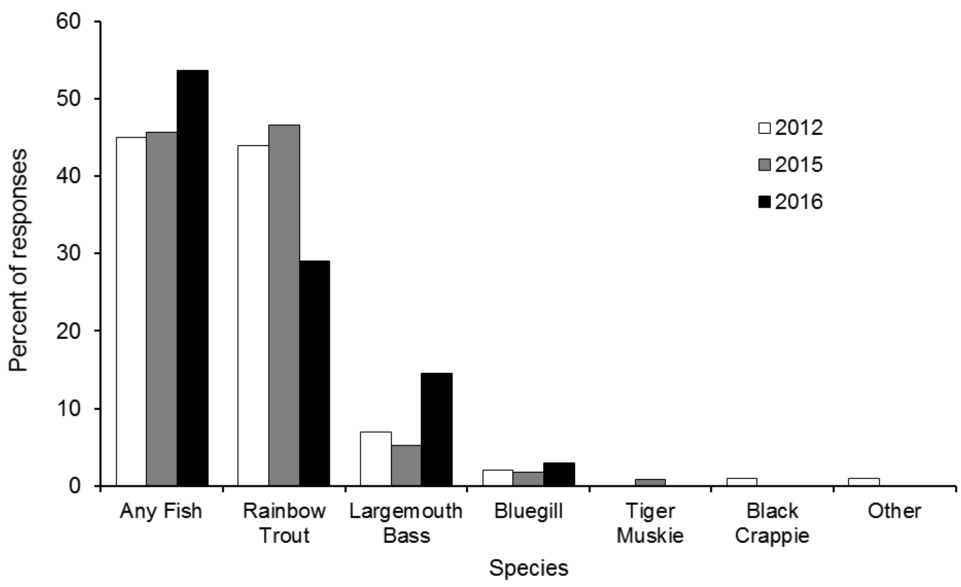


Figure 69. Comparison of angler responses collected during creel surveys regarding which fish species they were targeting at Spring Valley Reservoir, Idaho, in 2012, 2015, 2016.

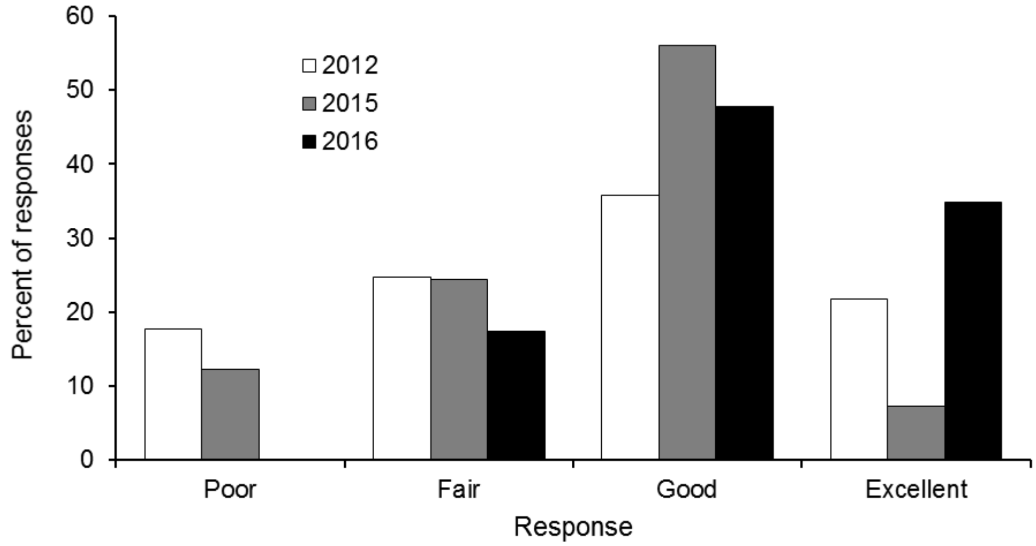


Figure 70. Comparison of angler responses collected during creel surveys regarding their overall fishing experience at Spring Valley Reservoir, Idaho, in 2012, 2015, and 2016.

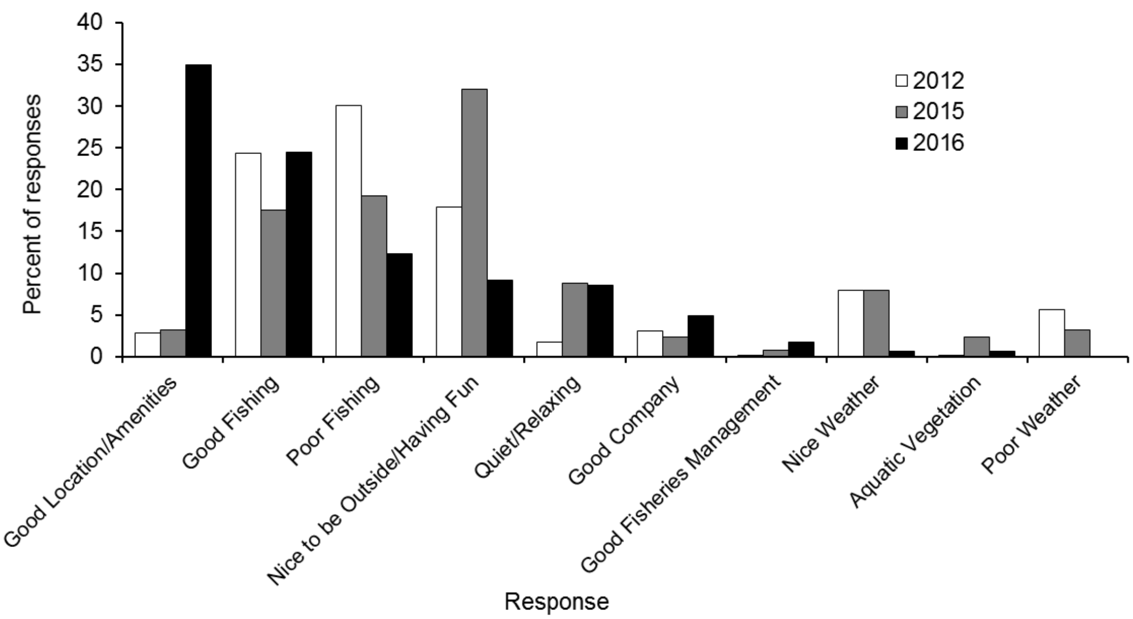


Figure 71. Comparison angler responses collected during creel surveys for what influenced the quality of their fishing experience when fishing at Spring Valley Reservoir, Idaho, in 2012, 2015 and 2016. (Only 10 most common answers shown).

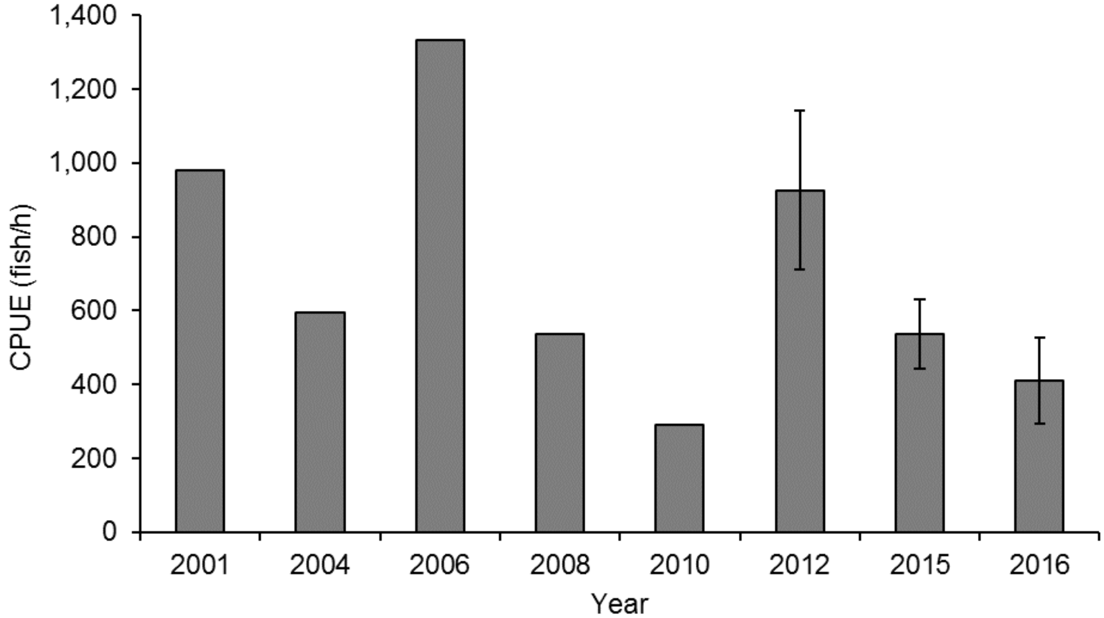


Figure 72. Comparison of catch-per-unit-effort (CPUE; fish/h) for fishes sampled through electrofishing in Spring Valley Reservoir, Idaho, from 2001 to 2016.

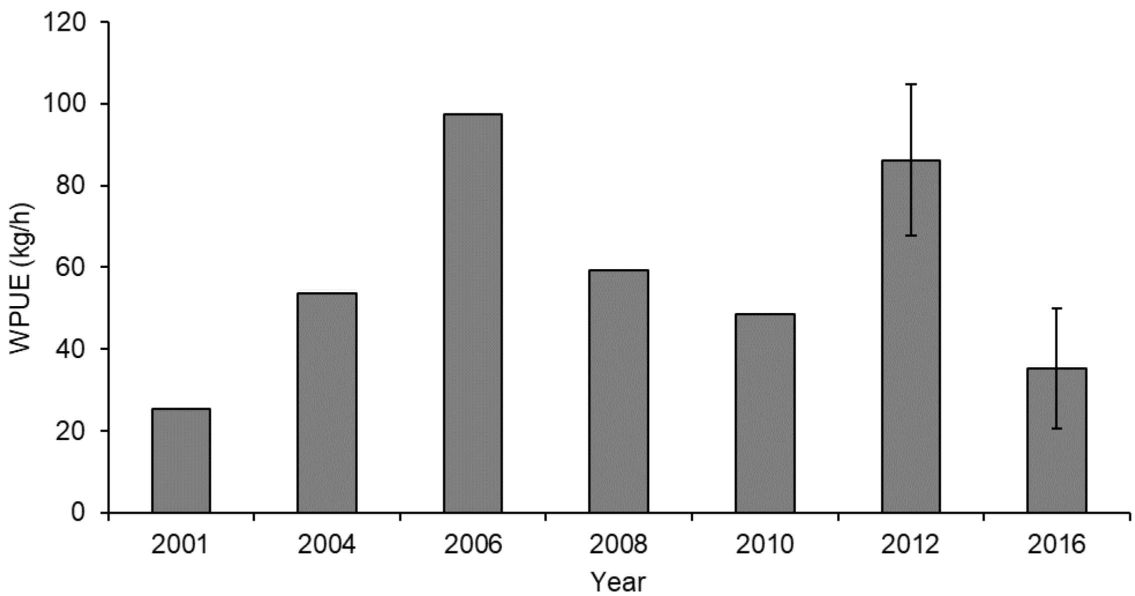


Figure 73. Weight-per-unit-effort (WPUE; kg/h) of fishes sampled by electrofishing Spring Valley Reservoir, Idaho, from 2001 to 2016. Error bars represent 90% confidence intervals.

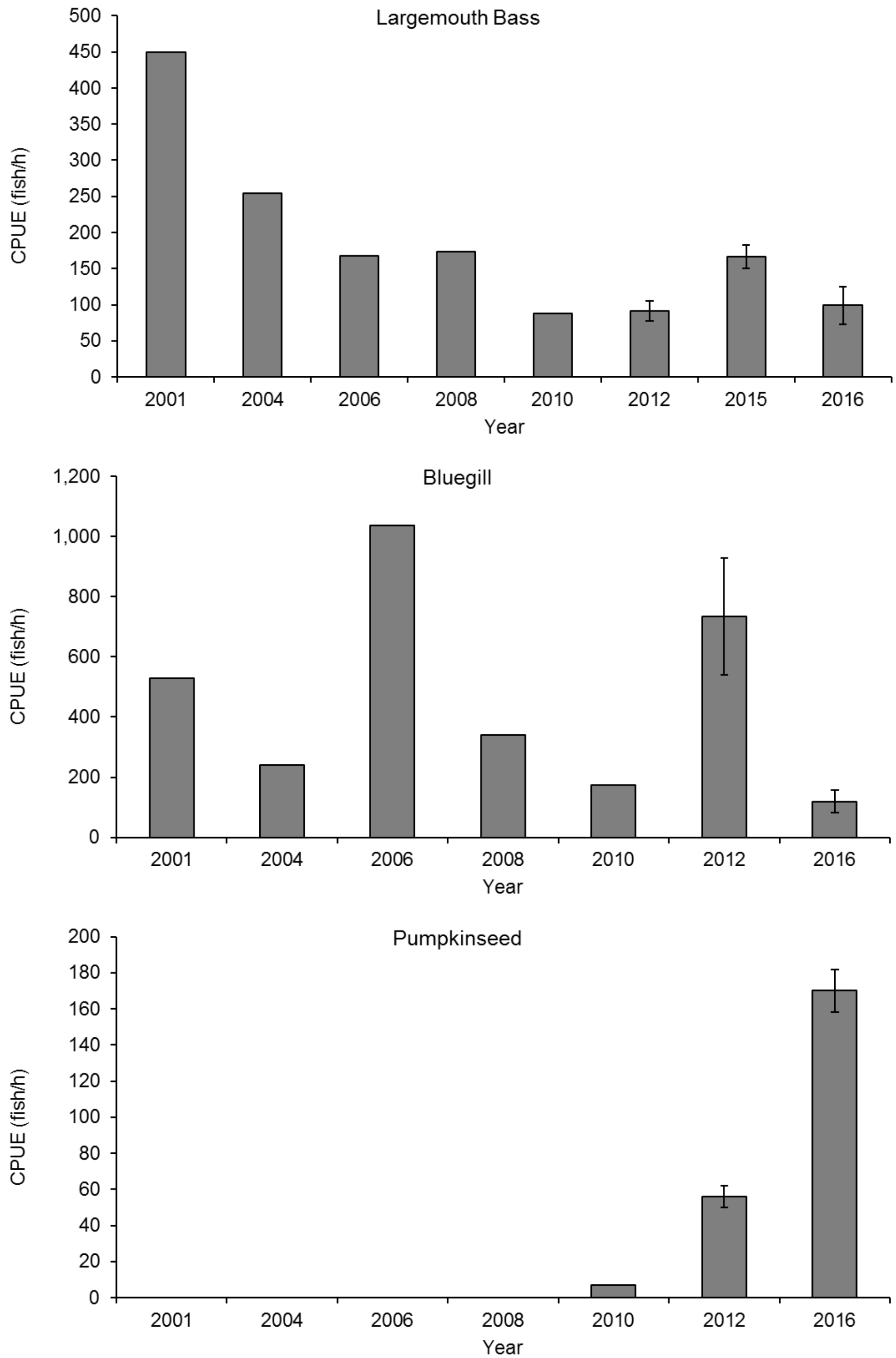


Figure 74. Catch-per-unit-effort (CPUE; fish/h), by species, of fishes sampled by electrofishing Spring Valley Reservoir, Idaho, from 2001 to 2016. Error bars represent 90% confidence intervals.

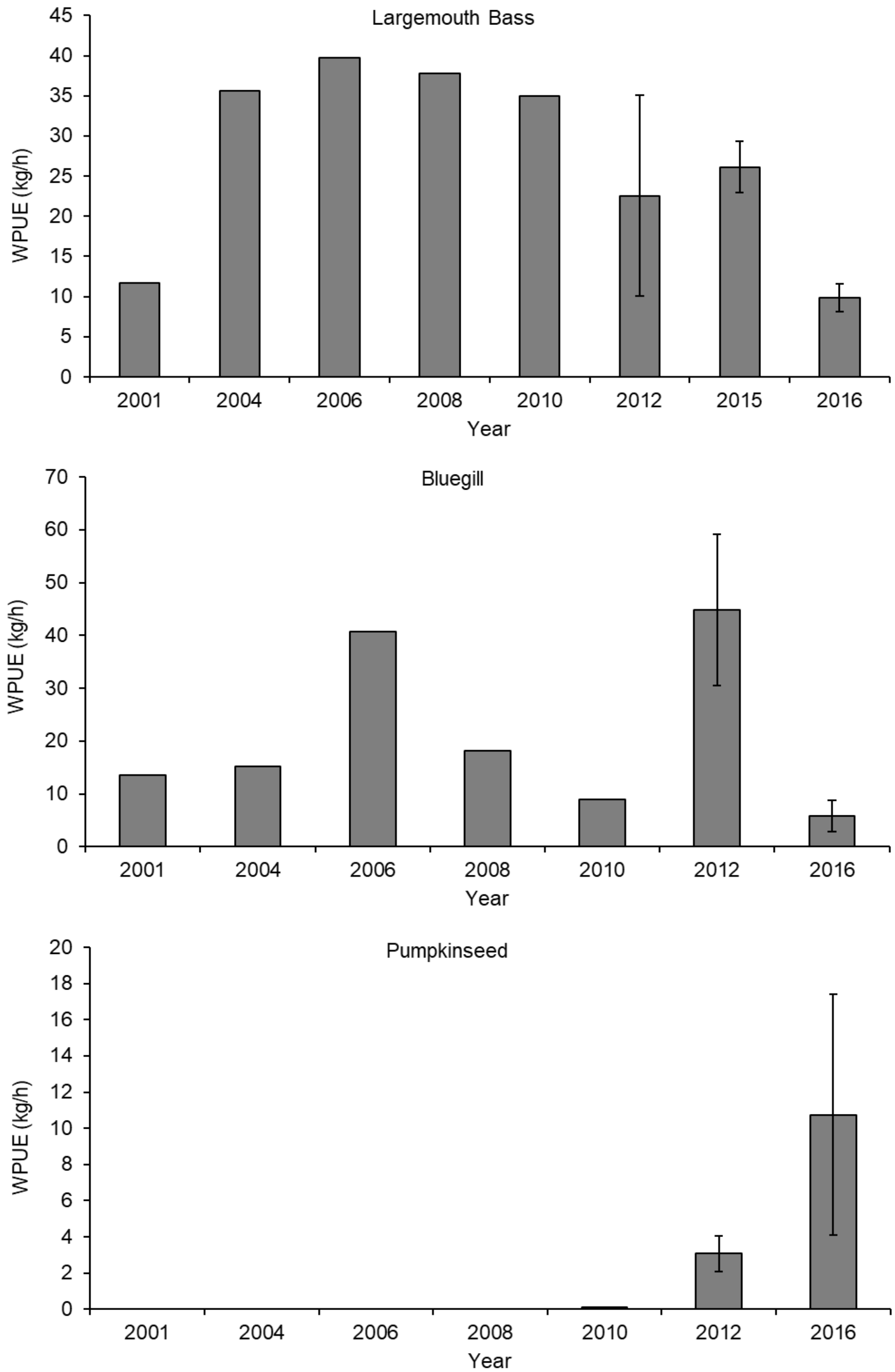


Figure 75. Weight-per-unit-effort (WPUE; kg/h), by species, of fishes sampled by electrofishing Spring Valley Reservoir, Idaho, from 2001 to 2016. Error bars represent 90% confidence intervals.

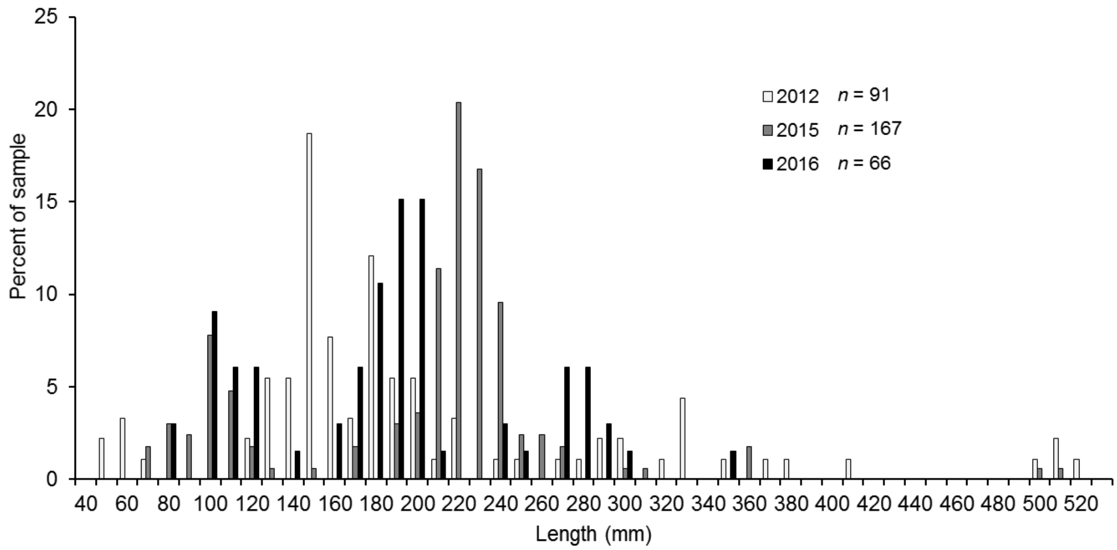


Figure 76. Comparison of Largemouth Bass length-frequency distributions from fish sampled through electrofishing in Spring Valley Reservoir, Idaho, in 2012, 2015 and 2016.

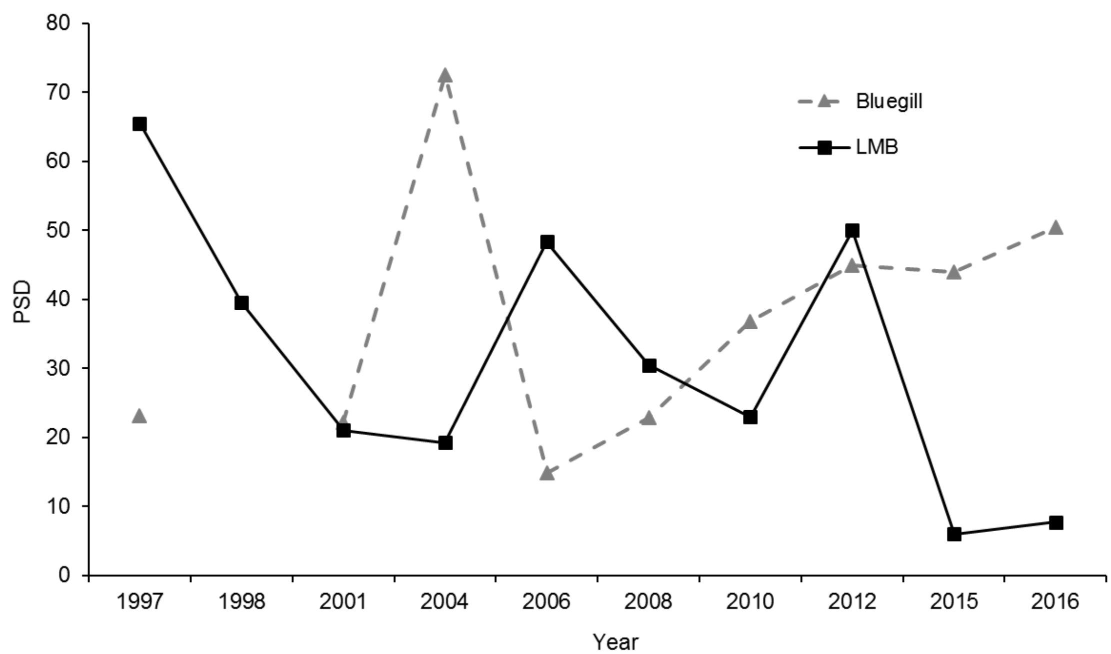


Figure 77. Proportional Size Distribution (PSD) values of Largemouth Bass and Bluegill sampled through electrofishing in Spring Valley Reservoir, Idaho, from 1997 to 2016.

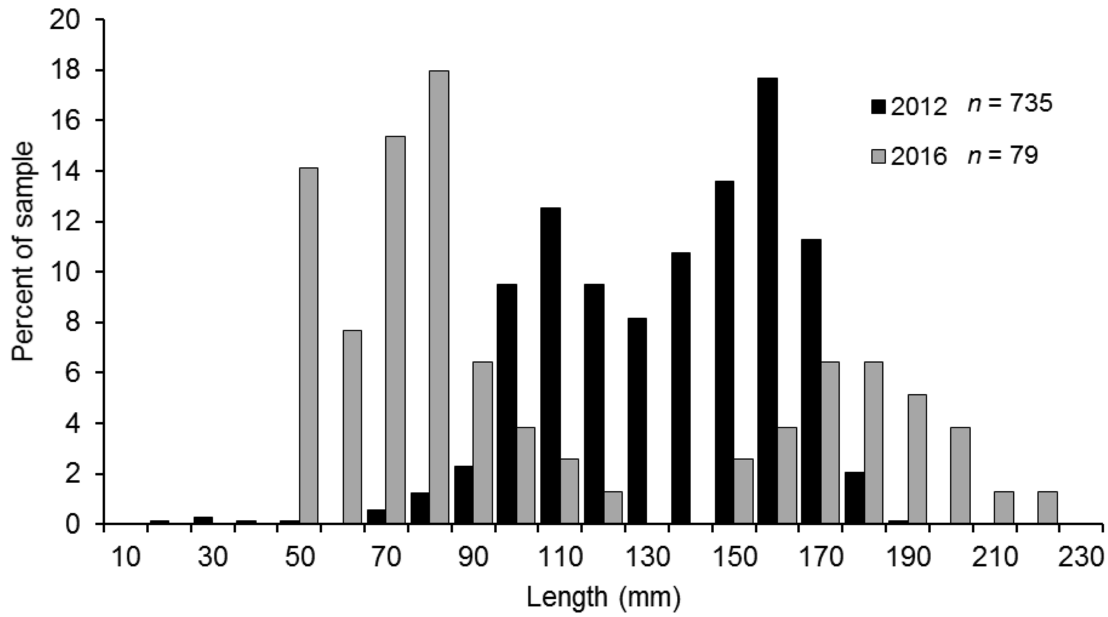


Figure 78. Comparison of Bluegill length-frequency distributions from fish sampled through electrofishing in Spring Valley Reservoir, Idaho, in 2012 and 2016.

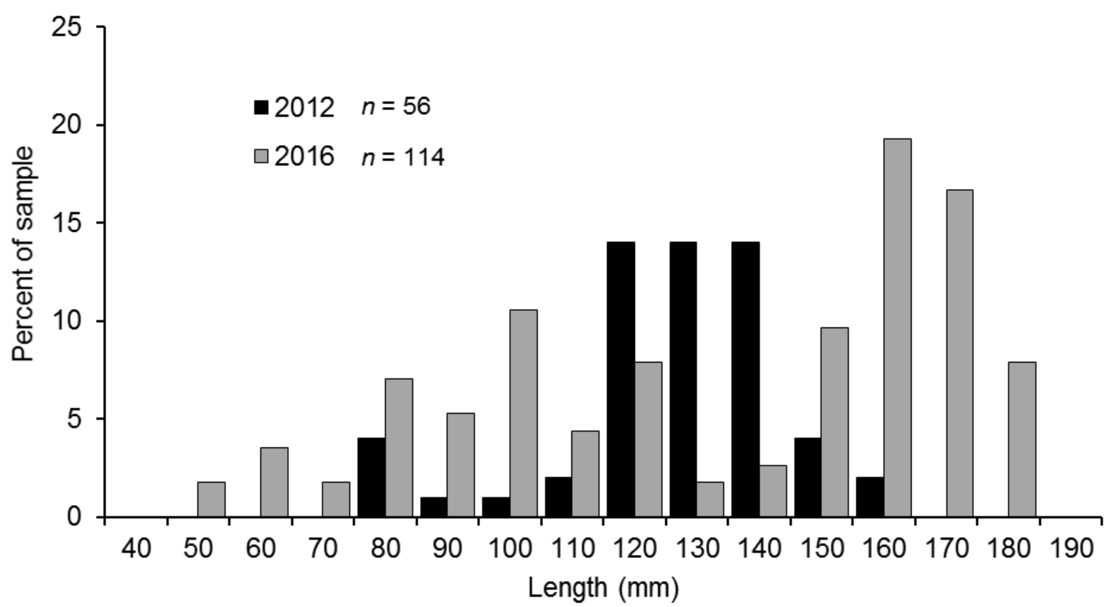


Figure 79. Comparison of Pumpkinseed length-frequency distributions from fish sampled through electrofishing in Spring Valley Reservoir, Idaho, in 2012 and 2016.

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# WAHA LAKE FISHERY EVALUATION


## ABSTRACT

In 2016, we evaluated Waha Lake to determine if it would be suitable for the reintroduction of a kokanee *Oncorhynchus nerka* fishery. Currently the lake is primarily managed as a “put-and-take” Rainbow Trout (RBT) *O. mykiss* fishery and is stocked multiple times per year with catchable-sized hatchery trout. While kokanee stocking has taken place in the past (most recently in 2009), and some of these fish continue to spawn annually, there is not enough natural production to maintain an adequate fishery. Thus, creating a viable kokanee fishery would require annual stockings. Our primary focus was the availability of adequate food resources, water temperature, and DO levels conducive for kokanee survival. Our sampling in 2016 found that the average length of *Daphnia sp.* (0.9 mm) was larger than the minimum size (>0.8 mm) preferred by kokanee, and average size was similar to the 0.90 mm average in 2012. Additionally, at least 33% of Daphnia in each sample were found to be at or above the preferred size, suggesting sufficient food resources for kokanee. Our sampling indicated the amount of available habitat preferred by kokanee was reduced during summer months due to unfavorable conditions. The maximum volume available for kokanee, as measured previously, was >1.6 M $m^3$. This is not concerning, as we have a successful kokanee fishery in nearby Soldiers Meadow Reservoir despite it having much lower volumes of water available for kokanee. With a large volume of the lake still available to kokanee, it does not appear that temperature or DO will be limiting factors. Previous sampling in 2008-2009 indicated that kokanee were numerous, but small in size, compared to 2016 where fish were larger but low in number. Stocking was previously curtailed in 2009 due to low return to creel. Our analysis suggests that stocking additional kokanee would result in reduced zooplankton length and numbers through additional predation. This would negatively impact growth and survival of stocked kokanee and trout, resulting in a poor trout fishery. Thus, we do not recommend renewed stocking of kokanee in Waha Lake



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## INTRODUCTION

Anglers have been asking for more kokanee *Oncorhynchus nerka* fishing opportunities in the Clearwater Region. In 2014, we began stocking kokanee in Soldier's Meadow Reservoir to address this need. This fishery is showing promise, so we began considering other opportunities to establish kokanee fisheries. Waha Lake has the most conducive temperature and dissolved oxygen (DO) profiles for kokanee of all our lowland lakes and reservoir. In fact, kokanee were previously stocked into Waha Lake in 20 years from 1982 to 2009 (Bowler and Schriever 1992; Cochnauer et al. 1996). These were primarily early-spawners, with late-spawners only stocked twice, in 1982 and 1997. Creel surveys indicated no kokanee were harvested in 1999 and only 26 were harvested in 2005 (Cochnauer et al. 2002; Hand et al. 2016). Based on this information, kokanee stocking in Waha Lake was discontinuing in 2009 (Hand 2009). Although stocking ended over a decade ago, a naturally reproducing population remains in the reservoir at low numbers. With the increased interest in kokanee fishing opportunities, we conducted fish, limnology, and zooplankton surveys in 2016 to evaluate the potential of renewed stocking of kokanee in Waha Lake.

## OBJECTIVES

1. Evaluate the potential of Waha Lake to support a kokanee fishery through renewed stocking efforts.

## STUDY AREA

Waha Lake is located approximately 30 km southeast of Lewiston, Idaho, and 2.5 km east of Waha, Idaho (Figure 1). At pool elevation of 1,032 m, it is a 36-ha, naturally-formed lake. It is the deepest of the Clearwater Region's lowland lakes, with a mean depth of 19.6 m and maximum depth of 33.0 m. It has a volume of 5.3M $m^3$ at full-pool elevation 1,032 m. The Lewiston Orchards Irrigation District (LOID) manages the water level at Waha Lake through pumping water into the lake from Soldier's Meadow Reservoir, and out of the lake to Mann Lake. As such, Waha Lake's pool level can fluctuate by as much as 11 m annually. The lake's primary inlet is located on the West Fork of Sweetwater Creek where LOID created a diversion dam. Normal full pool (generally around elevation 1,032 m, but up to 1,035 m in flood) is often not reached until early June in most years. Minimum pool (elevation 1,024 m) usually occurs in January when the water level has been reduced by outflow and pumping for summer irrigation (Hand et. al. 2012).

## METHODS

### Limnology

To provide information on habitat quality and quantity for kokanee, dissolved oxygen (DO) and temperature profiles, were conducted monthly from June to October. Dissolved oxygen and temperature profiles were taken from a boat with a YSI model 550A meter at the surface and 1-m increments down to at least 15 m. The boat was kept stationary in the deepest part of the lake while measurements were taken. Temperature was recorded in ºC, and dissolved oxygen in mg/L. This data was displayed over a depth profile. The volume of habitat available for kokanee was calculated as the total volume of water in Waha Lake where DO was >6.0 mg/L and temperature was <17 ºC. These values were selected based on preferences reported in Baldwin and Polacek (2002) and Berge (2009).

## Zooplankton

Monthly zooplankton sampling was conducted from June to October, 2016. Samples were collected with a Wisconsin-style plankton net (80-micron mesh, 30-cm diameter mouth). The boat was anchored at the deepest location on each lake based upon bathymetric maps and depth finder readings. When anchoring the boat, the anchor was slowly dropped and slack in the anchor line was let out to let the boat drift away from the anchor location. Three vertical tows were taken from that location. Tows were started 1.0 m above the bottom of the lake to avoid disturbing sediment. Depth of tow was recorded on each sample jar. Samples were rinsed into sample jars and stored in 70% ethyl alcohol. A Rite-in-the-Rain label was placed inside the sample jar. Samples were labeled with date, reservoir, number of tows, depth of tow, and personnel present.

Laboratory analysis was conducted based on a protocol developed previously for regional mountain lake surveys (Hand et al. 2016). Zooplankton samples were diluted into a known volume container (typically 100 ml) and 5 ml aliquots were then subsampled. Subsamples were counted until 200 of the most dominant families were observed. The density of zooplankton in each individual tow was then estimated by expanding the subsample estimate by total volume of the tow. Tow volume ($\pi$) was calculated by:

$$\pi \cdot r^2 \, x \, h$$

where $r$ = radius of the net and $h$ = depth of tow.

Zooplankton were counted based on three phylogenetic orders: Cladocera, Cyclopoida, and Calanoida. Within Cladocera (most common zooplankton), we identified individuals down to one of the following: Family Chydoridae, *Daphnia sp.*, *Ceriodaphnia sp.*, or *Bosmina sp.* In addition, the first 30 individuals of each category per sample were measured under the dissecting microscope to establish a length distribution for the sample.

## Population survey

Fish were sampled on October 20, 2016 using overnight (21.25 hours) gill net sets. Four gill nets were used, two floating and two sinking. Each measured 46 x 2 m, and were divided into six equal sized panels with bar mesh sizes of 20, 25, 32, 38, 51, and 64 mm. Sinking and floating nets were anchored on shore and set perpendicular to the shoreline. Nets were non-randomly spread throughout the reservoir (Figure 80) For the purpose of this evaluation, fish species, total lengths (mm), and weights (g) were recorded. Catch-per-unit-effort (CPUE) was calculated as the number of fish/net (individual net). Catch-per-unit-effort (CPUE; ± 90% confidence intervals) were calculated to compare with previous years. Significant differences in CPUE between years were determined to be those where 90% confidence intervals do not overlap. Mean length of fish (± 90% confidence intervals) were compared by species between years using standard two-sample *t*-tests (assuming equal variance) with a significance level of $\alpha = 0.05$.

# RESULTS

## Zooplankton

Zooplankton samples included five taxa of zooplankton: *Daphnia sp.*, *Ceriodaphnia sp.*, *Bosmina sp.*, Calanoida, and Cyclopoida. The sample composition of zooplankton throughout the field season were dominated by Cyclopoids, which ranged from a high of 87% of the individuals in June to a low of 60% in September. Daphnia sp., the preferred taxa for kokanee, represented as high as 25% of the composition in July and as low as 2% in October (Figure 81).

Zooplankton densities (# of individuals/$m^3$) were also highly variable (Figure 82). Bosmina (3,205/$m^3$) and Calanoida (1,336/$m^3$) densities peaked during August, but were substantially lower in other months. *Ceriodaphnia sp*. were only sampled during September and were otherwise absent from other months' samples. Cyclopoida showed the highest densities every month, peaking in August with a density of 7,310/$m^3$. Daphnia densities fluctuated throughout the season peaking in July with a density of 2,219/$m^3$ and recording a low of 39/$m^3$ in October.

Mean length of *Daphnia sp.* ranged from 0.78 to 1.06 mm, with the peak length in October (Figure 83). However, the small sample size ($n$ = 12) of *Daphnia sp.* collected during the October survey may not be representative of the population. The overall average length of 0.87 mm was similar to 2012 (0.90 mm; Hand et al. 2016). *Daphnia sp.* length frequencies from each sample show that the percent >0.8 mm ranged from 33 to 67% of each sample. Average lengths of Cyclopoida ranged from 0.27 to 0.40 mm, with average length peaking in August (Figure 84). Length-frequency distributions show that Cyclopoids >0.8 mm in length comprised <9% of the population in any sample (Figure 84).

## Limnology

During 2016, DO levels in the upper 8 m of the lake were generally above the 6.0 mg/L threshold considered stressful to kokanee (Figure 85). During our study, water temperatures in depths <3 m consistently exceeded the 17°C threshold preferred by kokanee (Figure 85). Based on these thresholds, the minimum volume of Waha Lake suitable for kokanee ranged from 1.6 M $m^3$ (30% of total volume) in June to a low of 0.9 M $m^3$ (17%) in August. It must be noted that the volumes for June and July are a minimum, as we were unable to sample temperature and DO below 20 m of depth due to limitations of our equipment. The hypolimnion occurred below 20 m during both months, thus the actual volume suitable for kokanee was larger than what we measured.

## Population survey

Gillnetting efforts resulted in the capture of 44 fish, including 15 kokanee, 18 Rainbow Trout (RBT) *Oncorhynchus mykiss*, 9 Yellow Perch (YP) *Perca flavescens*, and 2 splake *Salvelinus namaycush x Salvelinus fontinalis* (Table 16). The four nets had a combined CPUE of 3.8/net for kokanee, 4.5/net for RBT, 2.3/net for YP, and 0.5/net for splake. Kokanee ranged in length from 171 to 337 mm with an average of 251 mm (Figure 86). This average length was significantly larger ($P < 0.001$; $\alpha = 0.05$) than the 219 mm average length for surveys conducted in October 2009 while kokanee were still being stocked annually. In earlier studies (1989 - 1995), average lengths were similar, ranging from approximately 220 to 264 mm (Table 17; Bowler and Schriever 1992; Schriever and Cochnauer 1993; Cochnauer et al. 2001). Rainbow Trout ranged in length from 281 to 368 mm with an average of 326 mm (Figure 86). The two splake were 551 and 731 mm TL.

## DISCUSSION

Our primary determinants for renewed stocking of kokanee in Waha Lake were the availability of adequate food resources, water temperature and DO levels conducive for kokanee survival, and an evaluation of both current and historic fish population data. Kokanee are planktivores and prefer *Daphnia sp*., which puts them in competition with other species in the reservoir such as RBT and YP (Galbraith 1967; Beauchamp et al. 1995; Stark and Stockner 2006). Small zooplankton prey size may indicate cropping and little opportunity to support additional planktivorous fish species (Baldwin and Polacek 2002). Sampling in 2016 indicated that while average sizes of Cyclopoida taxa had declined compared to 2012, *Daphnia sp*. average size was similar (Hand et al. 2016). Additionally, the average length of *Daphnia sp.* (0.87 mm) was similar to the 0.90 mm average in 2012, and was larger than the minimum size (>0.80 mm) preferred by kokanee (Figure 83; Beauchamp et al. 1995; Stark and Stockner 2006; Hand et al. 2016). In 2016, at least 33% of *Daphnia sp.* in each sample were found to be at or above the preferred size, suggesting there are sufficient food resources for kokanee. However, densities of *Daphnia sp.* were lower than those found in other kokanee reservoirs such as Dworshak Reservoir (mean of 2,300 individuals/$m^3$ in non-restoration years; Wilson et al. 2016). This suggests that while there is currently an ample supply of preferred size food, there may not be a large enough zooplankton population to support additional stocking.

We were also concerned that although there might be enough zooplankton for kokanee, there might not be enough for other species. Kokanee are efficient consumers on zooplankton, to the point they can outcompete other fish for food resources. While *Oncorhynchus* species are known to feed on zooplankton down to 1.0 mm in length, they prefer individuals >1.3 mm (Galbraith 1975; Tabor et al. 1996; Wang et al. 1996). However, our surveys show that 11 - 67% of *Daphnia sp.* were >1.0 mm, and up to 17% of individuals were larger than the 1.3 mm preferred size. Overall, our data suggests that there is currently an adequate zooplankton population to support the populations of all planktivorous species already present in the lake. The primary issue with stocking kokanee in Waha Lake is the likelihood of reducing zooplankton size and numbers through additional predation. This could negatively impact growth and survival of stocked fish, resulting in a poor fishery.

Temperatures >17°C and DO levels <6.0 mg/L are considered stressful to both kokanee and RBT, and can result in reduced survival (Baldwin and Polacek 2002). Our sampling indicated that the amount of available habitat preferred by kokanee was reduced during summer months due to unfavorable conditions. The maximum volume available for kokanee, as measured previously, was at least 1.6 M $m^3$ (Figure 87). By October 2016, the amount of available habitat preferred by kokanee declined by 52% due to changes in temperature and DO levels (Figure 8). It must be taken into consideration that these calculations are assuming full pool conditions, and in the case that the lake is not at full pool conditions (e.g. from irrigation, evaporation, etc.) the amount of actual habitat available to kokanee during October may be even lower. While unfavorable temperature and DO conditions are problematic, kokanee are known to migrate throughout the water column in order to seek out favorable conditions (Baldwin and Polacek 2002). However, this is not concerning for Waha Lake, as we have a successful kokanee fishery in nearby Soldier's Meadow Reservoir despite it having much lower volumes of water available for kokanee (Hand et al. 2018). With a large volume of the lake still available to kokanee, it does not appear that temperature or DO are limiting factors.

Kokanee collected in 2016 included individuals that appeared to be mature (late spawners). Previous surveys indicated that kokanee were spawning up to age 4+, though most spawners were age 2-3 (Bowler and Schriever 1992; Schriever and Cochnauer 1993). A few kokanee still persist in the lake due to natural reproduction, although it has been inadequate to sustain a meaningful kokanee fishery. With the majority of the kokanee population consisting of early spawners, our sampling time was likely too late to properly document the entire population, as it occurred after the mature (age 2+) early spawners had likely spawned and died. We therefore recommend any future pelagic fish sampling be conducted in late August to ensure mature early spawner kokanee are properly sampled.

Previous pelagic surveys of Waha Lake were conducted in October 2008-2009 as part of a hatchery Rainbow Trout evaluation (Koenig and Meyer 2011). These surveys, conducted during the last years of kokanee stocking in Waha Lake, indicated large numbers of kokanee present in the lake, but at small sizes (219 mm average). In contrast, in 2016, catch rates were low and fish were larger. Although the mean length (251 mm) for kokanee in 2016 is within the “preferred” size range for anglers, the population is too small to attract anglers (Gablehouse 1984). With data from 2008-2009 indicating lower average lengths during previous stocking periods, renewed stocking will result in smaller kokanee. This would make the fishery unappealing to anglers while also affecting trout growth. Based on our re-evaluation, Waha Lake does not appear to be a good candidate for renewed stocking efforts in spite of the increased interest in kokanee fishing opportunities.

The capture of two splake in the survey is worth noting, as they have not been stocked since 1998. These fish were therefore at least 18 years old. Both fish had large heads in comparison to their thin bodies, suggesting a scarcity of desirable prey species. They are very long-lived so it is not surprising there are still individuals in the lake.

## MANAGEMENT RECOMMENDATIONS

1. We do not recommend renewed stocking of kokanee in Waha Lake due to the likelihood of depressing the size structure of both kokanee and Rainbow Trout populations.

Table 16. Number of fish caught and CPUE (fish/net) in gill nets in Waha Lake, Idaho, on October 20, 2016.

| | Number Caught | | | |
|---|---|---|---|---|
| | Sinking | Floating | Total | CPUE (90% CI) |
| Kokanee | 11 | 4 | 15 | 3.8 (±2.4) |
| Rainbow Trout | 6 | 12 | 18 | 4.5 (±1.6) |
| Yellow Perch | 0 | 9 | 9 | 2.3 (±2.2) |
| Splake | 1 | 1 | 2 | 0.5 (±0.5) |

Table 17. Summary of kokanee data collected during gill net surveys of Waha Lake, Idaho, from 1989 to 2016. *n* = number of fish collected; average length (mm); CPUE = catch-per-unit-effort (fish/net). Missing 90% confidence interval (C.I.) is due to lack of individual net data.

| Date | *n* | Average length (90% CI) | CPUE (90% CI) |
|---|---|---|---|
| September 1989[a] | 69 | 220* | --- |
| September 1997[b] | 249 | 264* | 62.3 |
| April 2009[c] | 134 | 227 (±7) | 16.8 (±5.9) |
| October 2009[c] | 446 | 215.7 (±3) | 37.2 (±6.4) |
| October 2016 | 15 | 251 (±20) | 3.8 (±2.4) |

*Individual fish lengths unavailable; average length estimated from length frequency distribution chart.

[a]Bowler and Schriever 1992

[b]Cochnauer et al. 2001

[c]Koenig and Meyer 2011


Figure 80. Location of four gill nets placed in Waha Lake, Idaho, on October 20, 2016. Floating nets are designated with an “F” label, and sinking nets with an “S” label.

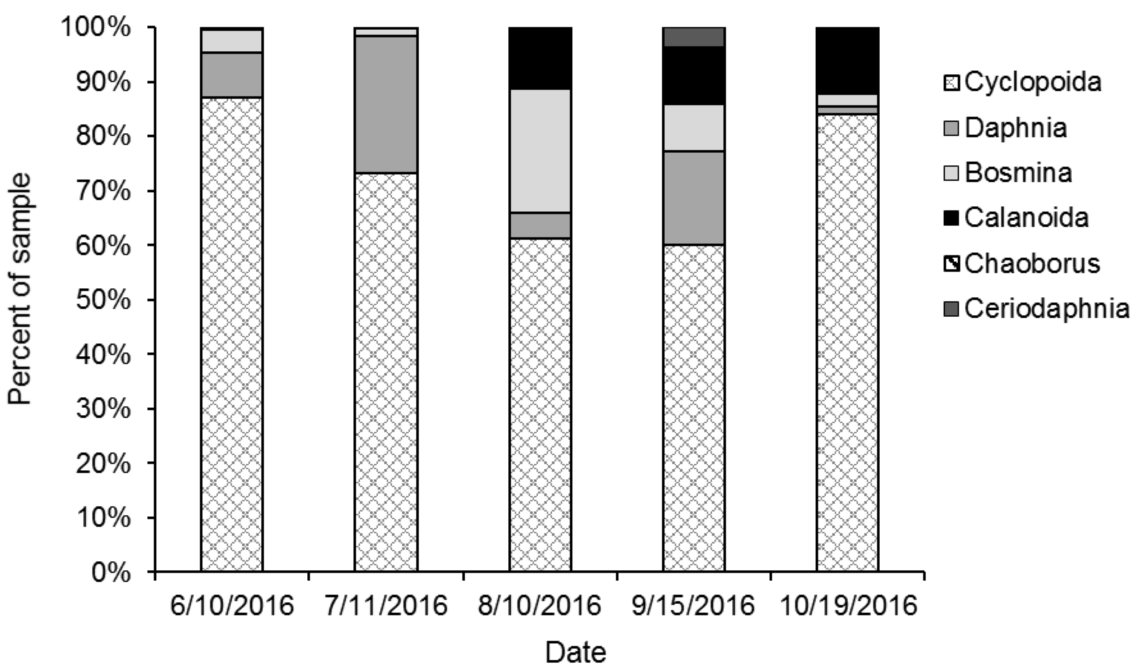


Figure 81. Monthly composition of zooplankton collected in Waha Lake, Idaho, during 2016.

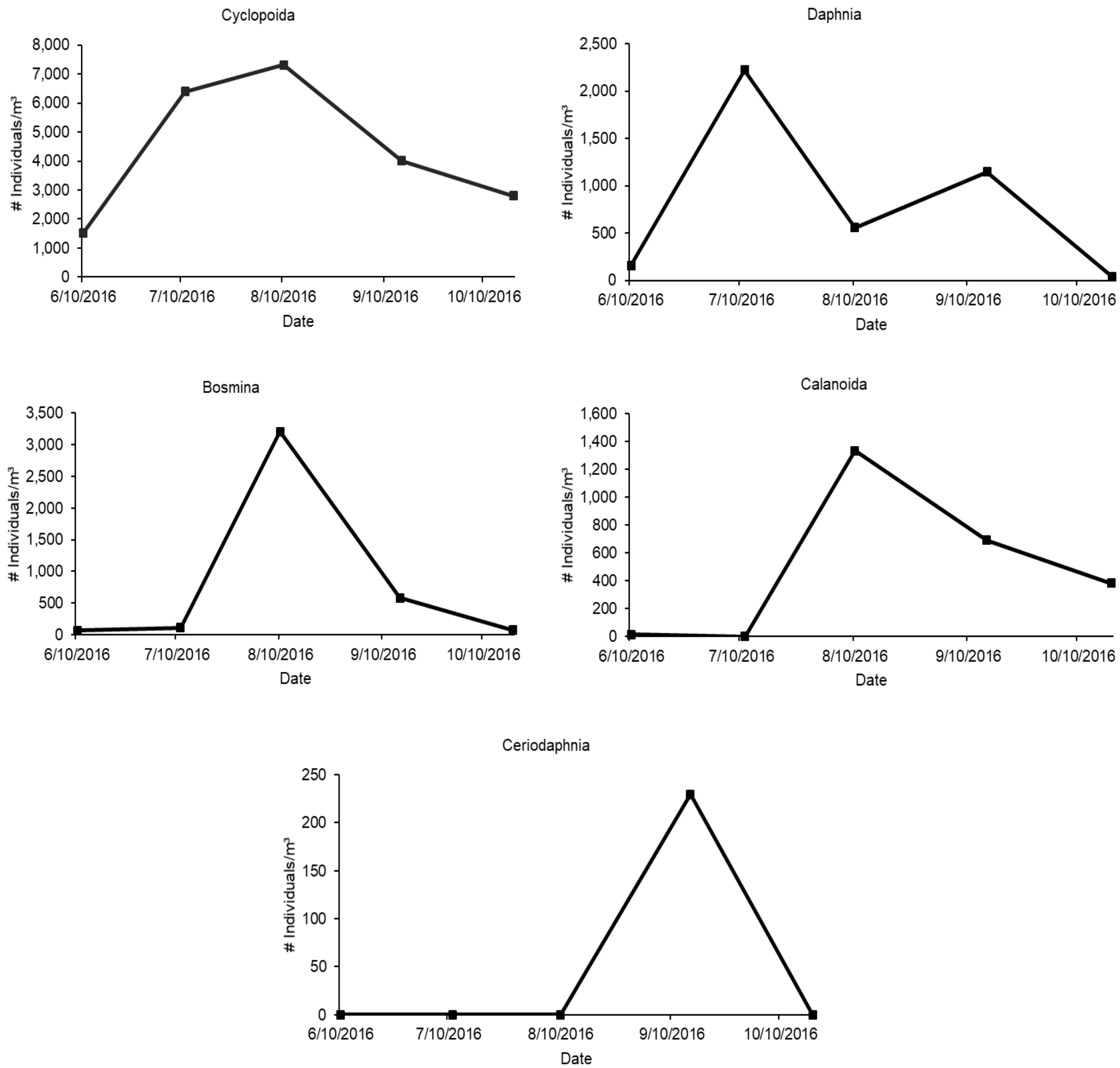


Figure 82. Monthly densities of zooplankton sampled in Waha Lake, Idaho, during 2016.

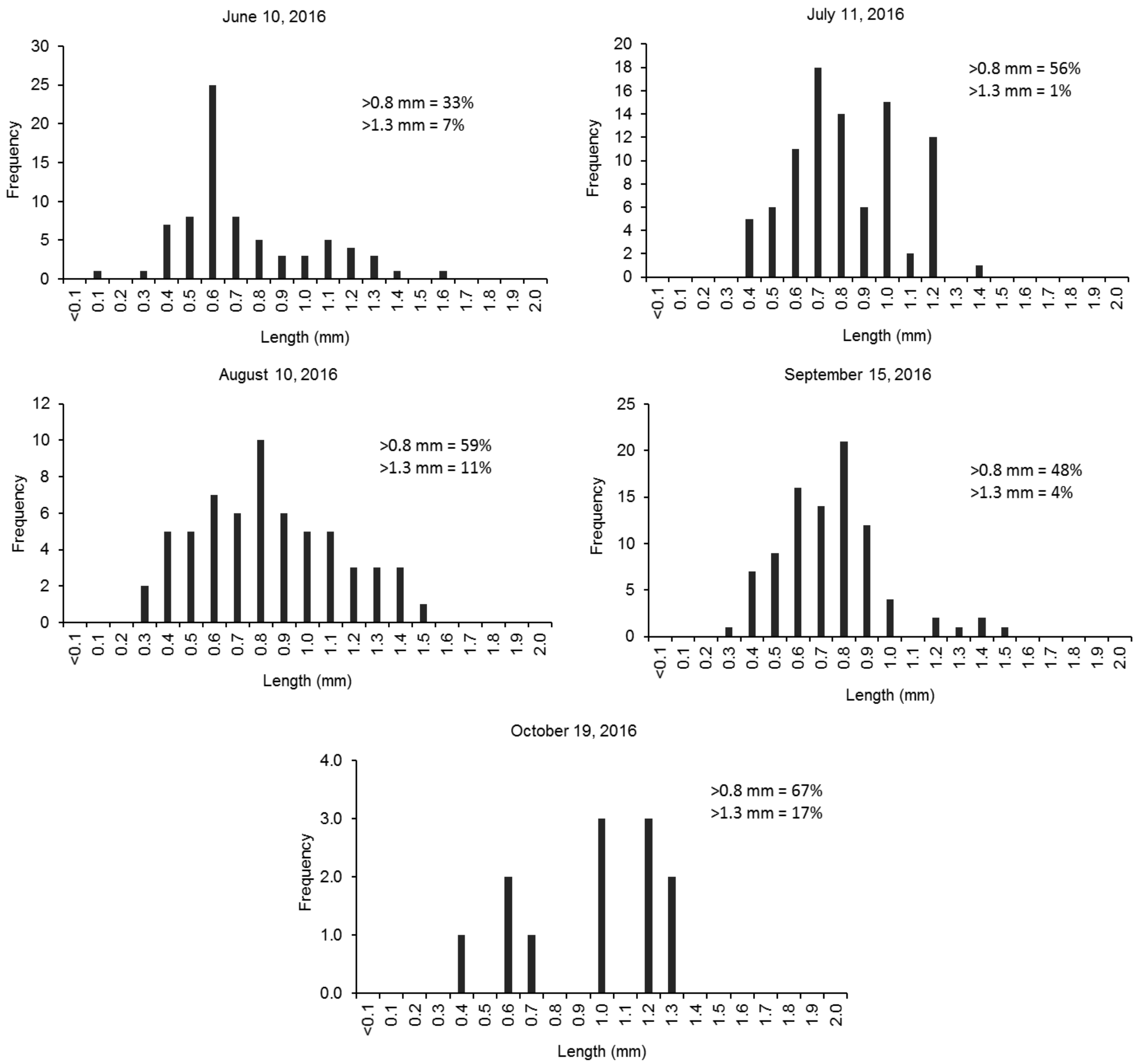


Figure 83. Length-frequency distributions of *Daphnia sp*. collected from Waha Lake, Idaho, during 2016.

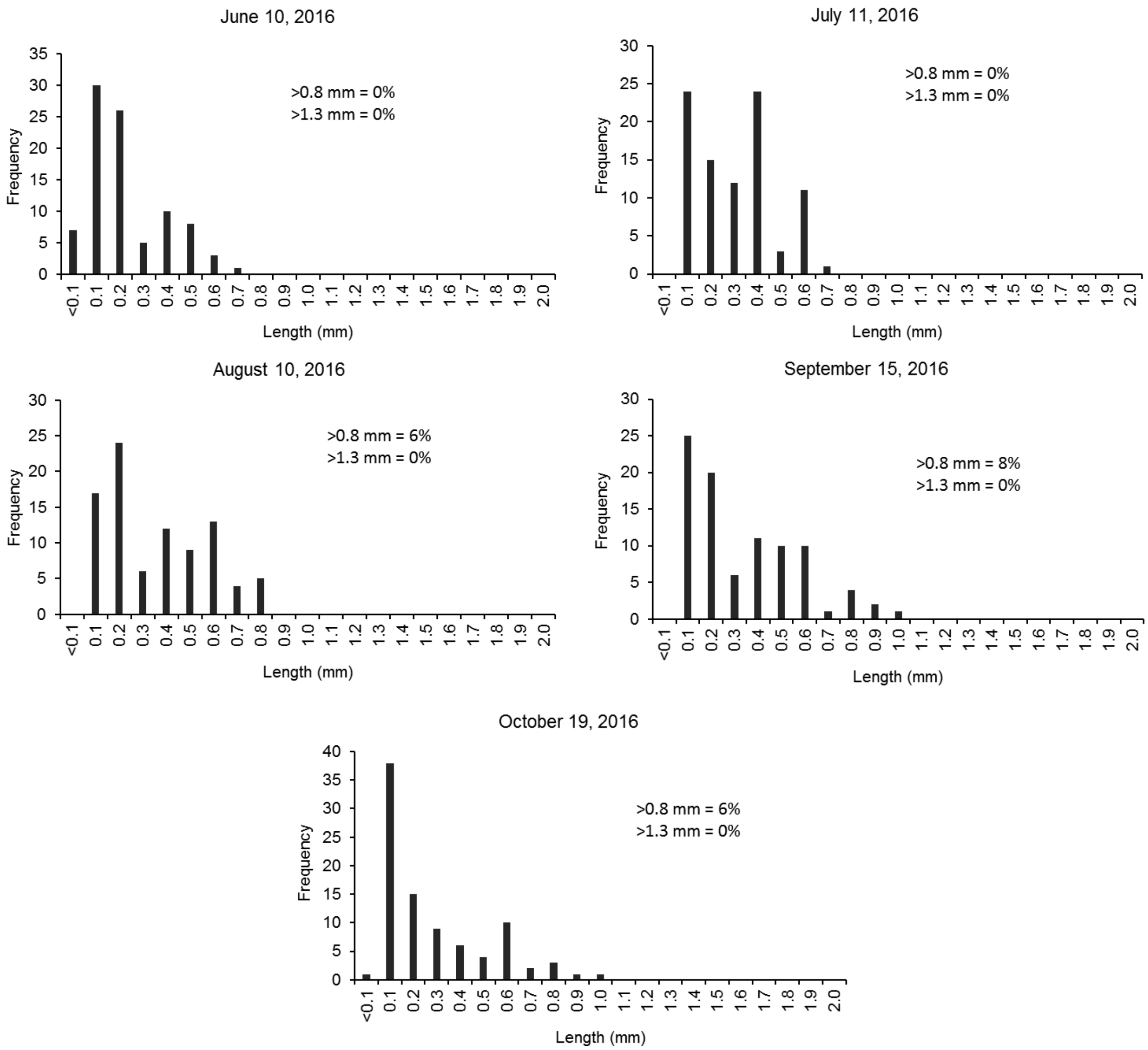


Figure 84. Length-frequency distributions of Cyclopoida collected from Waha Lake, Idaho, during 2016.

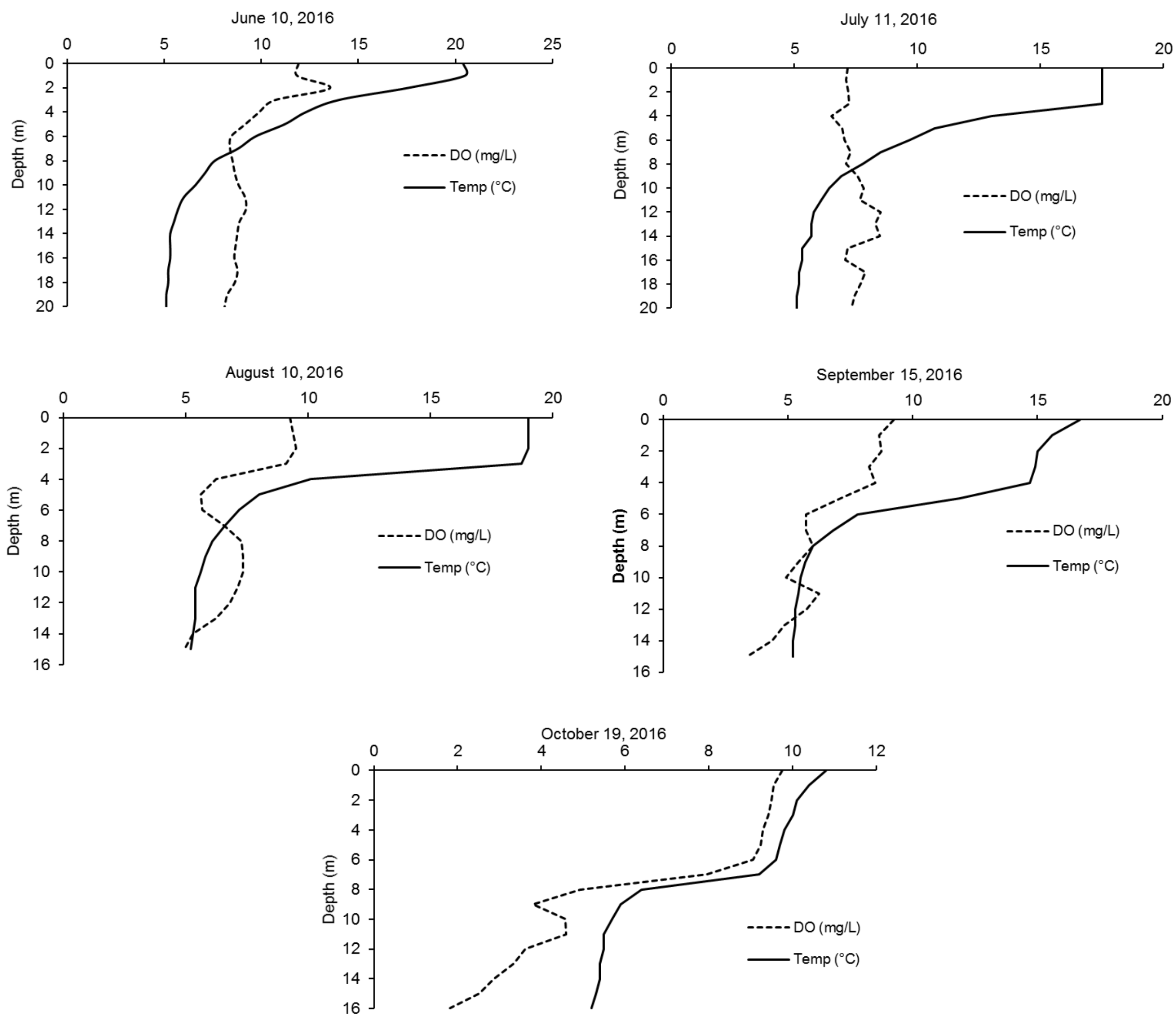


Figure 85. Monthly dissolved oxygen (DO mg/L) and temperature (°C) profiles for Waha Lake, Idaho, during 2016.

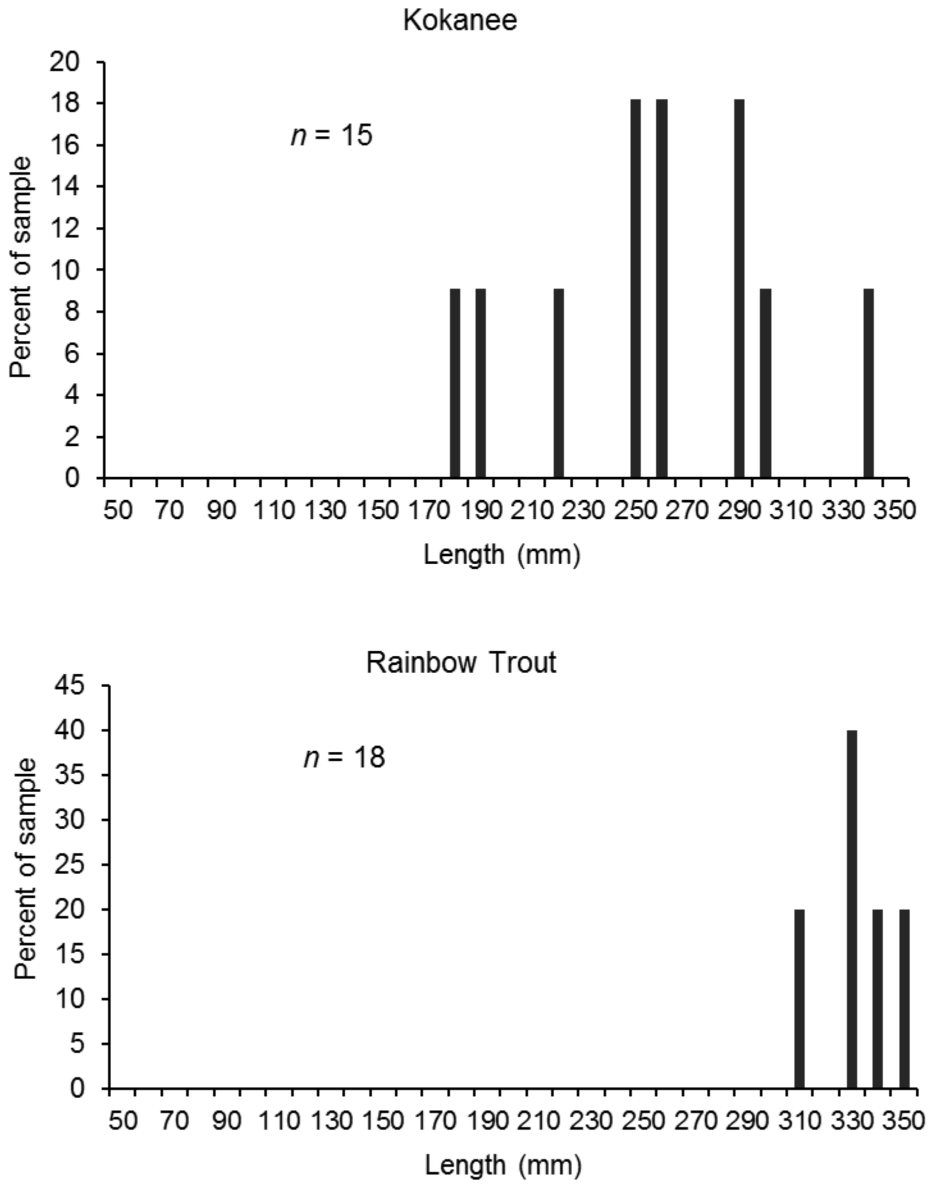


Figure 86. Length-frequency distributions of kokanee (top) and Rainbow Trout (bottom) caught by gill nets in Waha Lake, Idaho, in 2016.

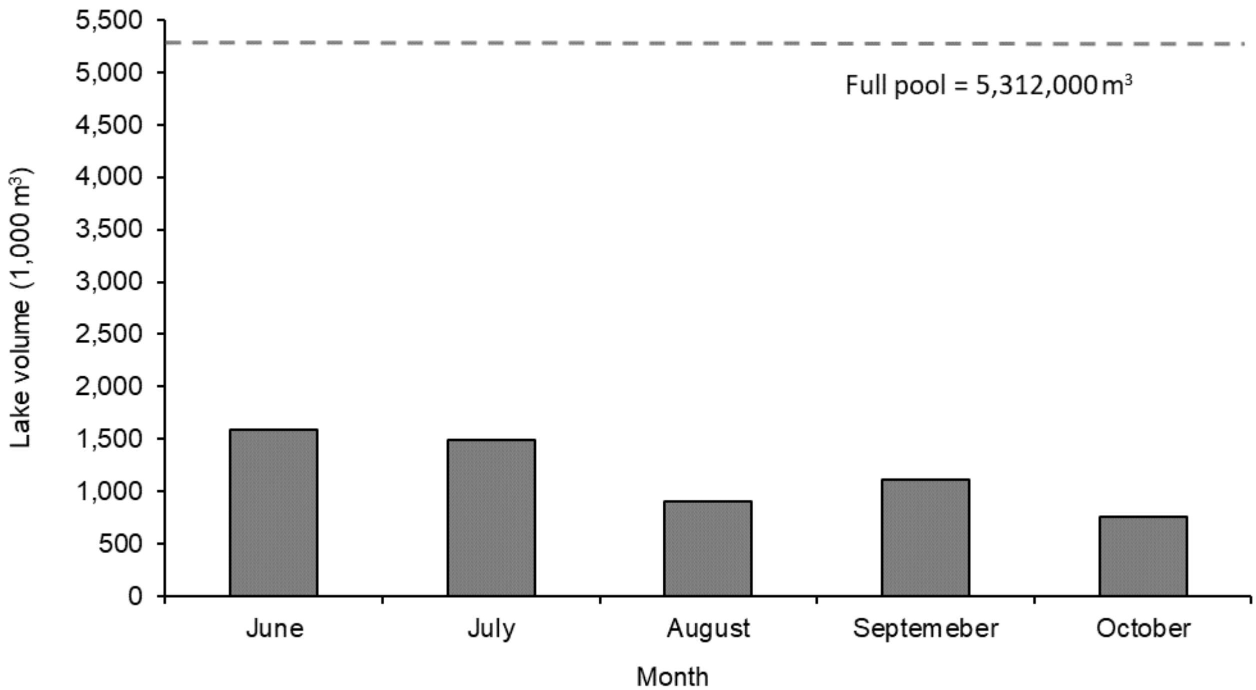


Figure 87. Volume of Waha Lake, Idaho, suitable for kokanee, based on dissolved oxygen and temperature measurements during 2016. Tolerances used: temperature (17°C) and dissolved oxygen (6.0 mg/L). *Volumes for June and July are a minimum, as we were unable to sample below 20 m of depth due to limitations of our equipment.

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## EVALUATION OF TROUT POPULATIONS IN THE NORTH FORK CLEARWATER RIVER


### ABSTRACT

Snorkel surveys were conducted on the main-stem North Fork Clearwater River (NFCR) and Kelly Creek in 2015 and 2016 to assess trends in trout populations. In 2016, abundances of Westslope Cutthroat Trout (WCT) *Oncorhynchus clarki lewisi* were 9.3/transect in Kelly Creek, and 10.2/transect in the NFCR. Compared to 2015, abundances in 2016 declined in Kelly Creek, and the NRCR below Cedars Campground. However, these declines were relatively small and are likely due to natural annual fluctuations, or possibly the high temperatures experienced in 2015. Overall, abundances continue to be substantially higher than was observed in the early 1970s, before the implementation of restrictive regulations and the construction of Dworshak Dam. In contrast to WCT trends, Rainbow Trout *O. mykiss* abundance in both Kelly Creek and the NFCR has generally declined since the 1970s. This decline is primarily attributable to the loss of the anadromous steelhead run in the NFCR due to the construction of Dworshak Dam. However, abundance in Kelly Creek has increased slightly since the low of 0.6/transect in 2011. Bull Trout *Salvelinus confluentus* were predominantly found in the NFCR compared to Kelly Creek during both 2015 and 2016. The vast majority of these fish (76% and 91%) were found in only two transects in the upper (roadless) section of the river. These two transects appear to serve as cool water refugia and staging areas prior to fall spawning. Mountain Whitefish *Prosopium williamsoni* were distributed throughout the drainage, while Smallmouth Bass *Micropterous dolomieu* were observed only in the lower reaches of the main stem NFCR.



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## INTRODUCTION

The North Fork Clearwater River (NFCR) and Kelly Creek (KC) are well known for their great fishing opportunities, drawing anglers from all over the country. Because of their popularity and the need to monitor population health, we conduct snorkel surveys in late July or early August every few years to monitor the native trout populations of these rivers. Surveys have been conducted since 1969 to look at trends in Westslope Cutthroat Trout (WCT) *Oncorhynchus clarki lewisi* and Rainbow Trout (RBT) *O. mykiss* abundance and distribution within the drainage. The initial studies were conducted to evaluate the status of WCT fisheries after declines in abundance were detected due to overharvest (Ball 1971; Johnson 1977). More recent surveys have been conducted to evaluate changes in abundance and size distribution since the implementation of restrictive regulations in the mid-1970s (Johnson 1977; Moffitt and Bjornn 1984; Hunt and Bjornn 1991). These studies established a set of snorkel survey transects throughout the drainage that we continue to use. In order to continue to track WCT abundance and distribution, we have initiated a four-year sampling rotation where we will surveys these historical transects for two consecutive years, then take two years off. This report will discuss the results of the surveys conducted in 2015 and 2016.

## OBJECTIVES

1. Assess trends in fish abundance, size, and distribution in the North Fork Clearwater River and Kelly Creek through snorkel surveys of historic monitoring transects.

## STUDY AREA

The North Fork Clearwater River is a sixth-order stream with a total drainage area of 739,983 ha Figure 88). The majority of the drainage is located on public lands and all snorkel transects were accessed off USFS Road 250. Kelly Creek is the largest tributary to the North Fork Clearwater River. Snorkel transects have been established since 1969 to assess WCT abundance and distribution within the main stem reach of the river (Hunt and Bjornn 1991). Historical transects were selected based upon fish holding capabilities, access, permanence for future studies, and included pools, runs, and pocket water habitats (Hunt and Bjornn 1991).

## METHODS

Snorkel surveys were conducted on the main-stem NFCR and Kelly Creek (KC) during July 20 - 30, 2015, and July 31 - August 4, 2016. (Figure 88). In 2015, a total of 48 transects were surveyed on the NFCR, while 35 transects were surveyed on KC. In 2016, the same 48 transects were surveyed on the NFCR, while the number of transects sampled on KC was reduced to 24 in order to complete the survey within a one-week survey window. Historic transects were located using GPS coordinates, pictures, and field notebooks from previous surveys. All NFCR and KC transect names were changed in 2016 to a more intuitive system (Table 18).

Snorkel surveys were conducted by one or two snorkelers, depending on the width of each transect. A single snorkeler was used only when the entire wetted width of the stream could be effectively observed by one person. The number of snorkelers surveying each transect was consistent with previous surveys to allow for direct comparison of data. Transects were snorkeled downstream, with each surveyor swimming close enough to shore to see the shoreline. Each snorkeler sampled towards the thalweg and towards their respective shorelines. All fish observed were counted, and length was estimated to the nearest inch for most species. For suckers, minnows, and sculpin, fish were categorized as > or < 305 mm. Transect length (m) and average width (m; based on five measurements) was measured using a rangefinder. Visibility (m) was estimated at each transect by having a snorkeler back away from a measuring tape until lettering on the tape was indistinguishable, the snorkeler then moved back towards the tape until the letters were viewable again. This distance was recorded to estimate visibility. Habitat type, date, time of day, water temperature, and weather conditions were also recorded for each transect. This report presents data on resident fish, while data for anadromous fish are presented in the 2016 Idaho Natural Production Monitoring and Evaluation (NPM) report (Stark et al. 2017).

Abundance was estimated using two methods in order to allow for comparisons with historic data. Historic surveys utilized a method of calculating density as the number of fish observed per transect. Our trend estimates also used this approach to allow for historical comparisons. We also estimated densities using area measurements to be consistent with modern estimates for snorkel surveys conducted in Idaho. These densities are displayed as "fish/100 $m^{2}$". Abundance (fish/transect, fish/100$m^2$; ± 90% confidence intervals) and size distributions were compared to previous surveys to evaluate trends. Significant differences in abundance between years were determined to be those where 90% confidence intervals do not overlap.

## RESULTS

### Kelly Creek

In 2015, the total fish count in KC was amongst the highest since monitoring began in 1969, with a total of 1,324 fish observed (Table 19). An average of 13.4 (±2.0) WCT were counted per transect. Westslope Cutthroat Trout were widely distributed throughout Kelly Creek with observations in all 35 transects sampled (Figure 2). Westslope Cutthroat Trout ranged in total length from 76 to 483 mm, with 44% of the fish >305 mm (Figure 3). An average of 1.6 RBT (±0.8) were observed per transect (Figure 89). Although few Bull Trout (BT) *Salvelinus confluentus* were observed, they were distributed throughout KC. However, 64% of those observed were in transect KC2 at the lower end of the creek. Mountain Whitefish (MWF) *Prosopium williamsoni* were very abundant (27.2/transect; ±6.9), and were found in 33 of the 35 transects (94%). Densities for KC were 0.62/100 $m^2$ (±0.14) for WCT, 0.11/100 $m^2$ (±0.06) for RBT, 0.23/100 $m^2$ (±0.23) for BT, and 2.37/100 $m^2$ (±0.96) for MWF.

In 2016, a total of 716 fish were observed during the survey (Table 20). An average of 9.3 (±3.3) WCT were counted per transect, and they were spatially distributed throughout KC with observations in 23 of the 24 (96%) transects sampled (Figure 89). The WCT observed ranged in total length from 102 to 482 mm, with 45% of the fish >305 mm (Figure 90). An average of 2.0 RBT (±1.7) were observed per transect (Figure 89). Three BT were observed in 2016, in transects KC6 and KC8 (Table 20). Mountain Whitefish were very abundant (19.5/transect; ±5.5), and were found in 21 of the 24 transects (88%). Fish densities for KC were 0.42/100 $m^2$ (±0.21) for WCT, 0.07/100 $m^2$ (±0.05) for RBT, <0.01/100 $m^2$ for BT, and 0.91/100 $m^2$ (±0.41) for MWF.

Westslope Cutthroat Trout densities in Kelly Creek peaked at 13.4/transect (±2.0) in 2015 for sampling from 1969 to 2016 (Figure 89). Although densities declined in 2016, they were still higher than any sampling conducted before 1989. Rainbow Trout densities increased in 2015 and

2016 after reaching their lowest level of 0.6/transect (±0.5) in 2010 (Figure 89), but still remain lower than those seen prior to 1973.

**North Fork Clearwater River**

In 2015, the total fish count in the NFCR was amongst the highest since monitoring began in 1969, with a total of 3,116 fish observed (Table 19). On average, 13.6 WCT (±3.0) were counted per transect, similar to that seen in Kelly Creek (Figure 91). They were widely distributed throughout NFCR with observations in 46 of the 48 transects (96%). Westslope Cutthroat Trout ranged in length from 51 to 483 mm, with 42% of the fish >305 mm (Figure 92). An average of 1.5 RBT (±0.5) were observed per transect (Figure 91). Bull Trout distribution was limited to the upper portion of the NFCR, from transect NF30 upstream (Table 19). Most of the BT (76%) were congregated in two transects (NF45 and NF48). Mountain Whitefish were very abundant (41.5/transect; ±12.5), and were observed in 45 of the 48 transects (94%). A large proportion (34%) of the MWF were observed in the five transects located from NF34 to NF38 (Table 19). Smallmouth Bass (SMB) *Micropterus dolomieu* were observed only in the lower section of the NFCR in five transects located from NF1 to NF6 (Table 19). Most of these fish (71%) were in transect NF1. Fish densities for the NFCR were 0.82/100 $m^2$ (±0.28) for WCT, 0.04/100 $m^2$ (±0.02) for RBT, 0.01/100 $m^2$ (±0.01) for BT, and 1.07/100 $m^2$ (±0.49) for MWF.

In 2016, the total number of fish counted in the NFCR declined by 34% from 2015, with a total of 2,044 fish observed (Table 20). On average, 10.1 WCT (±3.2) were observed per transect, slightly higher than seen in KC (Figure 91). They were spatially distributed throughout NFCR with observations in 44 of the 48 transects (92%). The WCT observed ranged in length from 51 to 432 mm, with 44% of the fish >305 mm (Figure 92). An average of 1.0 RBT (±0.5) were observed per transect (Figure 92). Bull Trout distribution was limited to the upper portion of the NFCR, from transect NF34 upstream (Table 20). Most of the BT (91%) were congregated in transects NF45 and NF48. Mountain Whitefish were very abundant (26.1/transect; ±12.5), and were observed in 43 of the 48 transects (90%). A large proportion (54%) of the MWF were observed in the five transects located from NF34 to NF38 (Table 20). Smallmouth Bass were observed only in the lower section of the NFCR in six transects located from NF1 to NF6 (Table 20). Fish densities for the NFCR were 0.66/100 $m^2$ (±0.26) for WCT, 0.05/100 $m^2$ (±0.04) for RBT, 0.71/100 $m^2$ (±0.77) for BT, and 1.53/100 $m^2$ (±0.80) for MWF.

Westslope Cutthroat Trout densities for the NFCR were similar to those observed in Kelly Creek, and reached their highest abundance (13.6/transect) in 2015 for sampling from 1969 to 2016 (Figure 91). Although densities declined in 2016, they were still the second highest observed since 1969. Densities followed similar patterns in all sections of the NFCR, except above the Cedars Campground, where densities increased to their highest level in 2016 (Figure 91). Rainbow Trout densities have been similar since 2011 for the NFCR as a whole, while remaining below levels observed in all sampling prior to 2011 (Figure 91). These trends were observed across all sections of NFCR, except above the Cedars Campground where no RBT were observed during sampling in 2015 or 2016. Smallmouth Bass (SMB) distribution was similar to previous surveys, and they have not been observed upstream of transect NF7.

## DISCUSSION

Over the course of snorkel surveys in the NFCR and Kelly Creek, trends in WCT and RBT abundance have changed substantially (Figure 89; Figure 91). Overharvest through the late 1960s and the construction of Dworshak Dam resulted in changes in fishing regulations in the early 1970s (Figure 93). Regulations prior to 1970 were “not more than 7 pounds and 1 fish, but not to exceed 15 fish”. Large declines in WCT populations led to new limits, including a three trout limit on the NFCR and catch and release on KC in 1970. Since then, Kelly Creek has remained catch and release, while the NFCR has had a few minor changes, most recently to a two trout limit with no WCT >356 mm in 2004.

Even though WCT densities in the NFCR and KC are now substantially higher than they were in the early 1970s, densities in this river system were lower than reported in most other northern Idaho rivers (Table 21). Although densities were lower than those in other north Idaho rivers, the proportion of WCT >355 mm (the legal length for harvest in the NFCR) has averaged 18% since 1989. This is similar to KC (20%), where regulations are catch and release, and the Lochsa River (20%) which has both catch and release and two fish >14” regulations. These proportions are all higher than those seen in the Selway (7%), which is only catch and release, and has limited access due to its wilderness location. These size structures are indicative of WCT populations with a low level of fishing-related mortality.

Over the last few surveys, we have observed some interesting trends in WCT abundance, including a decline in WCT abundance in KC since 1975, and across most of the NFCR in 2016 (Figure 89; Figure 91). A potential explanation for the decline in KC seen in 2016 was that we did not survey any transects above Box Creek due to time constraints. These transects historically have higher densities of WCT than the transects lower in the drainage. For example, in 2015, WCT densities averaged 0.62/100 $m^2$ in the lower transects, but averaged 2.26/100 $m^2$ in the upper transects. However, when comparing just the transects surveyed in both years, the declining trend persists, indicating that the differences in transects surveyed was not likely a causal factor this year. These transects were not surveyed due to time constraints in 2016, but the large difference in fish abundance observed historically between lower and upper transects suggests that we should make the effort to survey all of these transects in the future to provide a more complete picture. Another factor in these declines could be changes in fish distribution due to warmer water temperatures. Since these declines are appearing more in the lower reaches of the river, this would be plausible. Additionally, there has been an increase in WCT abundance in the upper NFCR above the Cedars Campground (Figure 91). One explanation is that fish are beginning to utilize this stretch of river more often due to warm water conditions. As the most upstream section of water we sample in this drainage, it contains the coolest water. If water temperatures rise due to climate change, this trend will likely continue in the future (Rahel and Olden 2008). Additionally, river flows in 2015-2016 were the lowest over the last 20 years (based on mean annual discharge). This may have resulted in all species being concentrated in smaller areas and thus easier to observe.

In contrast to WCT trends, RBT experienced a precipitous decline in densities beginning in 1972 (Figure 89; Figure 91). This decline in RBT abundance in the NFCR drainage has been attributed to the loss of juvenile steelhead after the construction of Dworshak Dam blocked adult steelhead (and all other anadromous fish) from returning to the NFCR (Pettit 1976; Moffitt and Bjornn 1984). More restrictive regulations for trout implemented in 1972 have not improved densities, indicating that anadromous steelhead were the source for local populations of resident RBT. In comparison, RBT densities have increased in the Lochsa and Selway rivers since the early 1970s when restrictive regulations were implemented (Hand et al. 2016). With the continued presence of anadromous fish in these rivers, this would suggest that the declines in RBT densities in the NFCR are likely due to the loss of anadromous fish in this system.

Bull Trout were predominantly observed in the NFCR compared to KC during both 2015 and 2016. This is similar data from previous snorkel surveys and radio tagging studies (Hand et al. 2013; Hansen et al. 2014). Interestingly, the vast majority of the BT observed in 2015 and 2016 (76% and 91%) were located in only two transects in the upper (roadless) section of the river. Bull Trout have been found to migrate out of Dworshak Reservoir and the lower reaches of the NFCR to spawn in tributaries during August - October (Hansen et al. 2014). These two transects, which are deeper run/pool habitat compared to surrounding habitat, appear to serve as cool water refugia and staging areas prior to fall spawning. This is expected, as BT are temperature sensitive and seek out cooler water temperatures in late summer (Rieman et al. 2007). Bull Trout densities have also been correlated with river flow conditions 3-4 years prior to sampling (Copeland and Meyer 2011). Flows in the NFCR during 2011-2012 were near the 20 year average, suggesting that the BT densities were more related to the factors mentioned above, as opposed to beneficial flow. However, as mentioned previously, flows in 2015-2016 were the lowest over the last 20 years. This may have resulted in BT being more concentrated in smaller areas of cool water refugia.

As with previous surveys, MWF were observed in high abundance throughout the NFCR drainage. The only trend data we have is for the NFCR below Weitas Creek, where MWF densities have ranged from 0.06 to 0.16/100 $m^2$ since 2011. Surveys conducted prior to 2011 did not include MWF information. These densities are much lower than KC and the upper NFCR, where we find the highest abundances of MWF in this drainage. This is to be expected due to cooler water in these areas.

Smallmouth Bass continue to be observed only in the lower reaches of the main-stem NFCR. There is concern that this introduced predator will move farther up into the drainage over time, especially if impacts of climate change cause water temperatures to increase (Rahel and Olden 2008). However, through 2016, we have not observed them upstream of transect NF7. We will continue to monitor their movement in the future through these snorkel surveys.

## MANAGEMENT RECOMMENDATIONS

1. Continue to monitor abundances, sizes and distribution of all fishes in the North Fork Clearwater River drainage on a two-year on, two-year off basis.

2. Survey all transects on Kelly Creek in the future to improve long-term trend comparisons.

Table 18. List of snorkel transects on the North Fork Clearwater River (NFCR) and Kelly Creek (KC), Idaho, including updated transect names, old transect names, and GPS coordinates.

| Stream | New site name | Old site name | Start latitude | Start longitude | End latitude | End longitude |
|---|---|---|---|---|---|---|
| NFCR | NF1 | NFCW1 | 46.84721 | 115.62598 | 46.84902 | 115.63020 |
| NFCR | NF2 | NFCW3 | 46.83816 | 115.57618 | 46.83764 | 115.57865 |
| NFCR | NF3 | NFCW4 | 46.83206 | 115.53732 | 46.83400 | 115.54398 |
| NFCR | NF4 | NFCW6 | 46.82646 | 115.48530 | 46.82726 | 115.48616 |
| NFCR | NF5 | NFCW8 | 46.82561 | 115.48266 | 46.82632 | 115.48432 |
| NFCR | NF6 | NFCW10 | 46.74237 | 115.54079 | 46.74375 | 115.53914 |
| NFCR | NF7 | NFCW11 | 46.73565 | 115.55088 | 46.73609 | 115.54916 |
| NFCR | NF8 | NFCW12 | 46.72861 | 115.55371 | 46.73110 | 115.55163 |
| NFCR | NF9 | NFCW13 | 46.72083 | 115.56434 | 46.72203 | 115.56449 |
| NFCR | NF10 | NFCW14 | 46.70621 | 115.55441 | 46.70798 | 115.55791 |
| NFCR | NF11 | NFCW15 | 46.69106 | 115.54861 | 46.69469 | 115.54900 |
| NFCR | NF12 | NFCW16 | 46.68386 | 115.54943 | 46.68626 | 115.54836 |
| NFCR | NF13 | NFCW18 | 46.67123 | 115.55138 | 46.67459 | 115.55296 |
| NFCR | NF14 | NFCW17 | 46.66095 | 115.54235 | 46.66202 | 115.54480 |
| NFCR | NF15 | NFCW19 | 46.63675 | 115.51347 | 46.64050 | 115.51676 |
| NFCR | NF16 | NFCW20 | 46.63192 | 115.50072 | 46.63182 | 115.50619 |
| NFCR | NF17 | NFCW21A | 46.62766 | 115.48219 | 46.62753 | 115.48342 |
| NFCR | NF18 | NFCW23 | 46.65580 | 115.39044 | 46.65500 | 115.39386 |
| NFCR | NF19 | NFCW24 | 46.68023 | 115.37244 | 46.67833 | 115.37200 |
| NFCR | NF20 | NFCW25 | 46.68785 | 115.35155 | 46.68580 | 115.35222 |
| NFCR | NF21 | NFCW27 | 46.69309 | 115.33360 | 46.69245 | 115.33471 |
| NFCR | NF22 | NFCW28 | 46.70907 | 115.32423 | 46.70874 | 115.32476 |
| NFCR | NF23 | NFCW29 | 46.72231 | 115.27605 | 46.72255 | 115.27812 |
| NFCR | NF24 | BC1 | 46.72401 | 115.25361 | 46.72409 | 115.25435 |
| NFCR | NF25 | BC2 | 46.72650 | 115.25040 | 46.72631 | 115.25099 |
| NFCR | NF26 | BC3 | 46.73584 | 115.24199 | 46.73563 | 115.24254 |
| NFCR | NF27 | BC4 | 46.75690 | 115.22977 | 46.75657 | 115.23058 |
| NFCR | NF28 | BC5 | 46.76980 | 115.22567 | 46.76929 | 115.22562 |
| NFCR | NF29 | BC6 | 46.78071 | 115.22043 | 46.78021 | 115.22145 |
| NFCR | NF30 | BC7 | 46.78799 | 115.21851 | 46.78742 | 115.21861 |
| NFCR | NF31 | BC8 | 46.79398 | 115.21603 | 46.79380 | 115.21646 |
| NFCR | NF32 | BC9 | 46.80366 | 115.21361 | 46.80308 | 115.21427 |
| NFCR | NF33 | BC10 | 46.81031 | 115.20818 | 46.80972 | 115.20842 |
| NFCR | NF34 | BCHC | 46.83152 | 115.17765 | 46.83088 | 115.17854 |
| NFCR | NF35 | BC11 | 46.83963 | 115.14635 | 46.83916 | 115.14665 |
| NFCR | NF36 | BC12 | 46.84014 | 115.13675 | 46.84044 | 115.13679 |
| NFCR | NF37 | BC13 | 46.84042 | 115.13171 | 46.84017 | 115.13236 |
| NFCR | NF38 | BC14 | 46.84100 | 115.10080 | 46.84113 | 115.10194 |

Table 18. (continued)

| Stream | New site name | Old site name | Start latitude | Start longitude | End latitude | End longitude |
|---|---|---|---|---|---|---|
| NFCR | NF39 | BC15 | 46.84701 | 115.09615 | 46.84701 | 115.09591 |
| NFCR | NF40 | BC16 | 46.86909 | 115.07954 | 46.86884 | 115.07990 |
| NFCR | NF41 | UC8 | 46.88487 | 115.09572 | 46.88477 | 115.09531 |
| NFCR | NF42 | UC7 | 46.89194 | 115.10393 | 46.89190 | 115.10342 |
| NFCR | NF43 | UC6 | 46.89598 | 115.11050 | 46.89553 | 115.11066 |
| NFCR | NF44 | UC5 | 46.90026 | 115.11692 | 46.90016 | 115.11652 |
| NFCR | NF45 | UC4 | 46.90082 | 115.11930 | 46.90065 | 115.11901 |
| NFCR | NF46 | UC3 | 46.90699 | 115.11655 | 46.90692 | 115.11672 |
| NFCR | NF47 | UC2 | 46.91449 | 115.11856 | 46.91434 | 115.11813 |
| NFCR | NF48 | UC1 | 46.91514 | 115.11995 | 46.91502 | 115.11957 |
| KC | KC1 | K1 | 46.71914 | 115.25065 | 46.71831 | 115.25098 |
| KC | KC2 | K2 | 46.72050 | 115.23937 | 46.72067 | 115.24021 |
| KC | KC3 | K3 | 46.71966 | 115.23222 | 46.71905 | 115.23340 |
| KC | KC4 | K4 | 46.72296 | 115.22765 | 46.72252 | 115.22922 |
| KC | KC5 | K5 | 46.72447 | 115.22427 | 46.72436 | 115.22514 |
| KC | KC6 | K6 | 46.72200 | 115.20679 | 46.72318 | 115.20768 |
| KC | KC7 | K7 | 46.71349 | 115.18475 | 46.71395 | 115.18536 |
| KC | KC8 | K8 | 46.71159 | 115.17695 | 46.71177 | 115.17807 |
| KC | KC9 | K9 | 46.71041 | 115.17397 | 46.71095 | 115.17500 |
| KC | KC10 | K10 | 46.70869 | 115.16689 | 46.70857 | 115.16836 |
| KC | KC11 | K11 | 46.71130 | 115.16132 | 46.71087 | 115.16179 |
| KC | KC12 | K12 | 46.71541 | 115.14783 | 46.71545 | 115.14943 |
| KC | KC13 | K13 | 46.71549 | 115.14432 | 46.71532 | 115.14538 |
| KC | KC14 | K14 | 46.71664 | 115.13882 | 46.71653 | 115.14046 |
| KC | KC15 | K15 | 46.71746 | 115.13559 | 46.71716 | 115.13654 |
| KC | KC16 | K17 | 46.70844 | 115.10785 | 46.70902 | 115.10839 |
| KC | KC17 | K20A | 46.70699 | 115.09204 | 46.70651 | 115.09203 |
| KC | KC18 | K19 | 46.71039 | 115.08949 | 46.70991 | 115.08922 |
| KC | KC19 | MID KC6 | 46.71946 | 115.08305 | 46.71991 | 115.08464 |
| KC | KC20 | MID KC5 | 46.71985 | 115.08069 | 46.71953 | 115.08161 |
| KC | KC21 | MID KC4 | 46.72236 | 115.07083 | 46.72197 | 115.07251 |
| KC | KC22 | MID KC3 | 46.72230 | 115.05978 | 46.72181 | 115.06058 |
| KC | KC23 | MID KC2 | 46.71736 | 115.05111 | 46.71717 | 115.05150 |
| KC | KC24 | MID KC1 | 46.71214 | 115.02850 | 46.71228 | 115.02968 |
| KC | KC25 | KC-6B | 46.71252 | 114.99527 | 46.71235 | 114.99583 |
| KC | KC26 | KC-6A | 46.71262 | 114.99458 | 46.71251 | 114.99509 |
| KC | KC27 | KC5 | 46.71458 | 114.99118 | 46.71454 | 114.99188 |
| KC | KC28 | KC4 | 46.71114 | 114.96329 | 46.71119 | 114.96389 |
| KC | KC29 | KC3 | 46.69481 | 114.92092 | 46.69522 | 114.92148 |
| KC | KC30 | KC2 | 46.69305 | 114.91714 | 46.69265 | 114.91727 |
| KC | KC31 | KC1 | 46.69640 | 114.90320 | 46.69599 | 114.90382 |

Table 19. Summary of fishes observed in snorkel transects on the North Fork Clearwater River (NFCR) and Kelly Creek (KC), Idaho, in 2015.

| New site name | Total length (m) | Average width (m) | Visability (m) | WCT | RBT | BT | MWF |
|---|---|---|---|---|---|---|---|
| NF1 | 494 | 49.0 | 2.7 | 11 | 5 | 0 | 10 |
| NF2 | 200 | 40.4 | 2.6 | 1 | 3 | 0 | 5 |
| NF3 | 550 | 48.2 | 2.7 | 11 | 4 | 0 | 5 |
| NF4 | 142 | 37.6 | 2.6 | 20 | 0 | 0 | 12 |
| NF5 | 193 | 36.8 | 2.6 | 1 | 2 | 0 | 18 |
| NF6 | 191 | 60.3 | 1.9 | 7 | 0 | 0 | 1 |
| NF7 | 188 | 35.4 | 1.9 | 4 | 4 | 0 | 0 |
| NF8 | 353 | 53.0 | 3.8 | 3 | 0 | 0 | 3 |
| NF9 | 160 | 44.8 | 2.6 | 0 | 0 | 0 | 11 |
| NF10 | 375 | 54.8 | 3.8 | 8 | 1 | 0 | 9 |
| NF11 | 436 | 51.6 | 3.8 | 0 | 0 | 0 | 5 |
| NF12 | 511 | 54.0 | 3.1 | 3 | 0 | 0 | 0 |
| NF13 | 450 | 40.4 | 3.1 | 20 | 0 | 0 | 24 |
| NF14 | 321 | 37.2 | 3.1 | 2 | 0 | 0 | 32 |
| NF15 | 500 | 34.6 | 1.9 | 17 | 0 | 0 | 83 |
| NF16 | 563 | 40.6 | 2.6 | 29 | 0 | 0 | 40 |
| NF17 | 109 | 29.8 | 2.6 | 3 | 0 | 0 | 9 |
| NF18 | 306 | 40.0 | 3.0 | 6 | 0 | 0 | 96 |
| NF19 | 206 | 52.0 | 3.0 | 5 | 0 | 0 | 14 |
| NF20 | 230 | 42.8 | 3.8 | 3 | 0 | 0 | 9 |
| NF21 | 120 | 49.6 | 3.9 | 7 | 0 | 0 | 34 |
| NF22 | 57 | 43.8 | 3.9 | 6 | 0 | 0 | 0 |
| NF23 | 149 | 52.2 | 3.8 | 13 | 0 | 0 | 30 |
| NF24 | 89 | 18.0 | 3.8 | 15 | 4 | 0 | 143 |
| NF25 | 53 | 21.4 | 3.8 | 6 | 5 | 0 | 69 |
| NF26 | 49 | 15.2 | 3.8 | 40 | 2 | 0 | 96 |
| NF27 | 84 | 19.4 | 3.8 | 30 | 6 | 0 | 100 |
| NF28 | 76 | 15.3 | 3.8 | 28 | 4 | 0 | 64 |
| NF29 | 112 | 18.0 | 3.8 | 13 | 5 | 0 | 90 |
| NF30 | 64 | 15.4 | 3.8 | 33 | 2 | 1 | 53 |
| NF31 | 46 | 20.0 | 3.8 | 14 | 5 | 0 | 25 |
| NF32 | 74 | 32.2 | 3.8 | 25 | 4 | 1 | 47 |
| NF33 | 65 | 20.8 | 3.8 | 9 | 3 | 0 | 42 |
| NF34 | 128 | 25.4 | 3.7 | 51 | 9 | 0 | 142 |
| NF35 | 65 | 29.6 | 3.7 | 21 | 5 | 0 | 10 |
| NF36 | 32 | 23.0 | 3.7 | 34 | 1 | 1 | 153 |
| NF37 | 60 | 24.0 | 3.7 | 48 | 0 | 0 | 200 |
| NF38 | 116 | 25.8 | 3.7 | 28 | 0 | 4 | 200 |

Table 19. (continued)

| New site name | Total length (m) | Average width (m) | Visability (m) | WCT | RBT | BT | MWF |
|---|---|---|---|---|---|---|---|
| NF39 | 72 | 20.0 | 3.7 | 7 | 0 | 2 | 50 |
| NF40 | 48 | 24.6 | 3.7 | 9 | 0 | 2 | 22 |
| NF41 | 44 | 22.0 | 4.5 | 4 | 0 | 0 | 1 |
| NF42 | 43 | 16.4 | 4.5 | 5 | 0 | 0 | 2 |
| NF43 | 53 | 15.2 | 4.5 | 1 | 0 | 1 | 2 |
| NF44 | 28 | 16.2 | 3.2 | 7 | 0 | 37 | 4 |
| NF45 | 49 | 15.6 | 3.2 | 11 | 0 | 150 | 14 |
| NF46 | 28 | 12.6 | 3.2 | 7 | 0 | 0 | 5 |
| NF47 | 52 | 13.2 | 4.5 | 2 | 0 | 21 | 2 |
| NF48 | 35 | 13.0 | 4.5 | 4 | 0 | 72 | 4 |
| KC1 | 110 | 36.4 | 2.1 | 6 | 2 | 0 | 1 |
| KC2 | 59 | 29.4 | 2.1 | 16 | 11 | 0 | 22 |
| KC3 | 122 | 33.8 | 2.1 | 23 | 7 | 1 | 15 |
| KC4 | 123 | 31.6 | 1.9 | 14 | 0 | 0 | 13 |
| KC5 | 69 | 29.2 | 2.1 | 12 | 4 | 0 | 39 |
| KC6 | 140 | 20.6 | 1.9 | 15 | 4 | 0 | 38 |
| KC7 | 60 | 39.2 | 1.9 | 11 | 3 | 1 | 22 |
| KC8 | 74 | 31.8 | 2.1 | 21 | 2 | 0 | 14 |
| KC9 | 114 | 25.8 | 1.9 | 25 | 1 | 0 | 28 |
| KC10 | 118 | 25.2 | 2.1 | 29 | 3 | 0 | 67 |
| KC11 | 59 | 25.4 | 4.9 | 14 | 6 | 0 | 30 |
| KC12 | 112 | 35.2 | 4.9 | 20 | 1 | 0 | 14 |
| KC13 | 91 | 38.2 | 4.9 | 4 | 3 | 0 | 18 |
| KC14 | 113 | 23.2 | 4.9 | 14 | 1 | 0 | 43 |
| KC15 | 87 | 24.8 | 4.9 | 6 | 0 | 0 | 48 |
| KC16 | 73 | 30.4 | 4.9 | 23 | 3 | 0 | 37 |
| KC17 | 40 | 22.3 | 4.9 | 12 | 0 | 0 | 85 |
| KC18 | 62 | 14.3 | 4.9 | 16 | 0 | 0 | 65 |
| KC19 | 160 | 31.2 | 3.5 | 22 | 3 | 1 | 65 |
| KC20 | 95 | 35.8 | 3.5 | 3 | 0 | 0 | 0 |
| KC21 | 180 | 36.0 | 4.0 | 17 | 1 | 0 | 15 |
| KC22 | 73 | 20.0 | 4.0 | 8 | 0 | 0 | 0 |
| KC23 | 47 | 26.0 | 4.0 | 7 | 0 | 0 | 61 |
| KC24 | 72 | 29.4 | 4.5 | 4 | 0 | 0 | 48 |
| KC25 | 61 | 20.0 | 3.5 | 9 | 0 | 0 | 3 |
| KC26 | 37 | 16.0 | 3.5 | 6 | 0 | 1 | 20 |
| KC27 | 54 | 21.3 | 3.5 | 14 | 0 | 0 | 4 |
| KC28 | 10 | 15.0 | 4.5 | 5 | 0 | 0 | 5 |
| KC29 | 53 | 13.5 | 4.5 | 12 | 0 | 0 | 10 |
| KC30 | 43 | 12.0 | 4.5 | 9 | 0 | 7 | 4 |
| KC31 | 23 | 13.5 | 4.5 | 19 | 0 | 0 | 8 |

Table 20. Summary of fishes observed in snorkel transects on the North Fork Clearwater River (NFCR) and Kelly Creek (KC), Idaho, in 2016.

| New site name | Total length (m) | Average width (m) | Visability (m) | WCT | RBT | BT | MWF |
|---|---|---|---|---|---|---|---|
| NF1 | 494 | 57.0 | 2.6 | 7 | 0 | 0 | 2 |
| NF2 | 200 | 44.5 | 2.6 | 6 | 0 | 0 | 4 |
| NF3 | 550 | 55.0 | 2.2 | 7 | 0 | 0 | 0 |
| NF4 | 142 | 38.5 | 2.7 | 16 | 0 | 0 | 2 |
| NF5 | 193 | 40.8 | 2.7 | 2 | 0 | 0 | 1 |
| NF6 | 191 | 53.0 | 2.6 | 3 | 0 | 0 | 2 |
| NF7 | 188 | 38.0 | 2.9 | 1 | 6 | 0 | 1 |
| NF8 | 306 | 42.0 | 2.7 | 10 | 3 | 0 | 3 |
| NF9 | 140 | 45.6 | 2.7 | 0 | 3 | 0 | 4 |
| NF10 | 382 | 57.5 | 1.7 | 3 | 0 | 0 | 15 |
| NF11 | 436 | 47.8 | 2.8 | 1 | 0 | 0 | 4 |
| NF12 | 300 | 43.8 | 2.4 | 2 | 1 | 0 | 1 |
| NF13 | 450 | 41.0 | 2.4 | 2 | 0 | 0 | 6 |
| NF14 | 188 | 41.4 | 2.8 | 2 | 0 | 0 | 0 |
| NF15 | 500 | 38.0 | 3.0 | 0 | 0 | 0 | 18 |
| NF16 | 560 | 36.2 | 2.8 | 2 | 7 | 0 | 7 |
| NF17 | 108 | 35.0 | 3.0 | 2 | 1 | 0 | 0 |
| NF18 | 306 | 39.3 | 2.8 | 8 | 0 | 0 | 60 |
| NF19 | 206 | 54.0 | 2.8 | 4 | 0 | 0 | 3 |
| NF20 | 228 | 43.6 | 2.1 | 2 | 0 | 0 | 20 |
| NF21 | 128 | 51.2 | 2.1 | 6 | 0 | 0 | 4 |
| NF22 | 57 | 44.0 | 2.9 | 1 | 0 | 0 | 0 |
| NF23 | 150 | 49.8 | 2.9 | 6 | 0 | 0 | 18 |
| NF24 | 89 | 18.5 | 2.8 | 3 | 0 | 0 | 35 |
| NF25 | 53 | 20.6 | 2.4 | 1 | 0 | 0 | 24 |
| NF26 | 49 | 17.4 | 2.4 | 0 | 0 | 0 | 21 |
| NF27 | 84 | 19.2 | 2.0 | 4 | 2 | 0 | 20 |
| NF28 | 76 | 23.9 | 3.2 | 14 | 4 | 0 | 10 |
| NF29 | 112 | 25.6 | 3.9 | 3 | 2 | 0 | 15 |
| NF30 | 64 | 17.3 | 4.0 | 17 | 12 | 0 | 36 |
| NF31 | 46 | 22.5 | 4.0 | 8 | 1 | 0 | 41 |
| NF32 | 74 | 34.2 | 3.6 | 27 | 0 | 0 | 61 |
| NF33 | 65 | 24.0 | 4.1 | 43 | 3 | 0 | 25 |
| NF34 | 130 | 27.4 | 3.2 | 45 | 2 | 2 | 90 |
| NF35 | 63 | 29.8 | 4.1 | 10 | 0 | 0 | 3 |
| NF36 | 32 | 22.8 | 3.7 | 36 | 0 | 3 | 76 |
| NF37 | 59 | 23.0 | 3.7 | 56 | 0 | 1 | 258 |
| NF38 | 112 | 23.6 | 4.1 | 40 | 1 | 1 | 250 |

Table 20. (continued)

| New site name | Total length (m) | Average width (m) | Visability (m) | WCT | RBT | BT | MWF |
|---|---|---|---|---|---|---|---|
| NF39 | 74 | 22.0 | 4.6 | 23 | 1 | 1 | 58 |
| NF40 | 51 | 26.8 | 3.5 | 2 | 0 | 1 | 6 |
| NF41 | 45 | 27.0 | 5.6 | 16 | 0 | 0 | 30 |
| NF42 | 45 | 12.4 | 5.5 | 2 | 0 | 0 | 4 |
| NF43 | 50 | 15.0 | 5.4 | 0 | 0 | 3 | 1 |
| NF44 | 49 | 15.0 | 5.7 | 10 | 0 | 8 | 5 |
| NF45 | 49 | 16.4 | 5.8 | 21 | 0 | 149 | 4 |
| NF46 | 26 | 12.8 | 5.2 | 7 | 0 | 0 | 1 |
| NF47 | 50 | 12.2 | 5.9 | 1 | 0 | 1 | 7 |
| NF48 | 35 | 11.6 | 4.2 | 6 | 0 | 53 | 0 |
| KC1 | 125 | 41.2 | 5.1 | 3 | 2 | 0 | 9 |
| KC2 | 62 | 29.4 | 5.1 | 3 | 3 | 0 | 16 |
| KC3 | 132 | 37.0 | 5.5 | 5 | 4 | 0 | 3 |
| KC4 | 120 | 33.8 | 6.0 | 3 | 1 | 0 | 3 |
| KC5 | 72 | 29.0 | 4.8 | 1 | 0 | 0 | 21 |
| KC6 | 153 | 24.8 | 5.4 | 4 | 3 | 1 | 45 |
| KC7 | 75 | 35.4 | 4.2 | 1 | 0 | 0 | 26 |
| KC8 | 74 | 33.8 | 4.6 | 6 | 0 | 0 | 8 |
| KC9 | 115 | 30.8 | 4.5 | 17 | 4 | 2 | 33 |
| KC10 | 116 | 28.6 | 5.5 | 26 | 24 | 0 | 0 |
| KC11 | 59 | 27.2 | 4.6 | 11 | 0 | 0 | 14 |
| KC12 | 112 | 36.6 | 5.3 | 5 | 0 | 0 | 33 |
| KC13 | 93 | 40.6 | 5.3 | 0 | 1 | 0 | 14 |
| KC14 | 115 | 24.4 | 5.0 | 1 | 1 | 0 | 42 |
| KC15 | 87 | 25.4 | 6.0 | 13 | 0 | 0 | 31 |
| KC16 | 73 | 32.0 | 4.2 | 23 | 0 | 0 | 0 |
| KC17 | 46 | 25.6 | 5.2 | 4 | 3 | 0 | 36 |
| KC18 | --- | --- | --- | | | | |
| KC19 | 190 | 29.2 | 4.4 | 8 | 0 | 0 | 10 |
| KC20 | 95 | 34.2 | 4.4 | 1 | 0 | 0 | 0 |
| KC21 | 180 | 37.6 | 4.4 | 21 | 0 | 0 | 24 |
| KC22 | 73 | 17.4 | 4.4 | 3 | 0 | 0 | 1 |
| KC23 | 47 | 22.0 | 4.4 | 28 | 0 | 0 | 55 |
| KC24 | 72 | 26.6 | 4.3 | 28 | 3 | 0 | 25 |
| KC25 | --- | --- | --- | --- | --- | --- | --- |
| KC26 | --- | --- | --- | --- | --- | --- | --- |
| KC27 | --- | --- | --- | --- | --- | --- | --- |
| KC28 | --- | --- | --- | --- | --- | --- | --- |
| KC29 | --- | --- | --- | --- | --- | --- | --- |
| KC30 | --- | --- | --- | --- | --- | --- | --- |
| KC31 | --- | --- | --- | --- | --- | --- | --- |

Table 21. Comparison of Westslope Cutthroat Trout densities (fish/100 $m^2$) in northern Idaho rivers.

| River | Year | Density |
|---|---|---|
| St. Joe River | 2013 | 1.42 |
| Selway River | 2015 | 1.00 |
| N.F. Coeur d'Alene River | 2013 | 0.75 |
| N.F. Clearwater River | 2016 | 0.66 |
| Kelly Creek | 2016 | 0.42 |
| Locha River | 2013 | 0.32 |

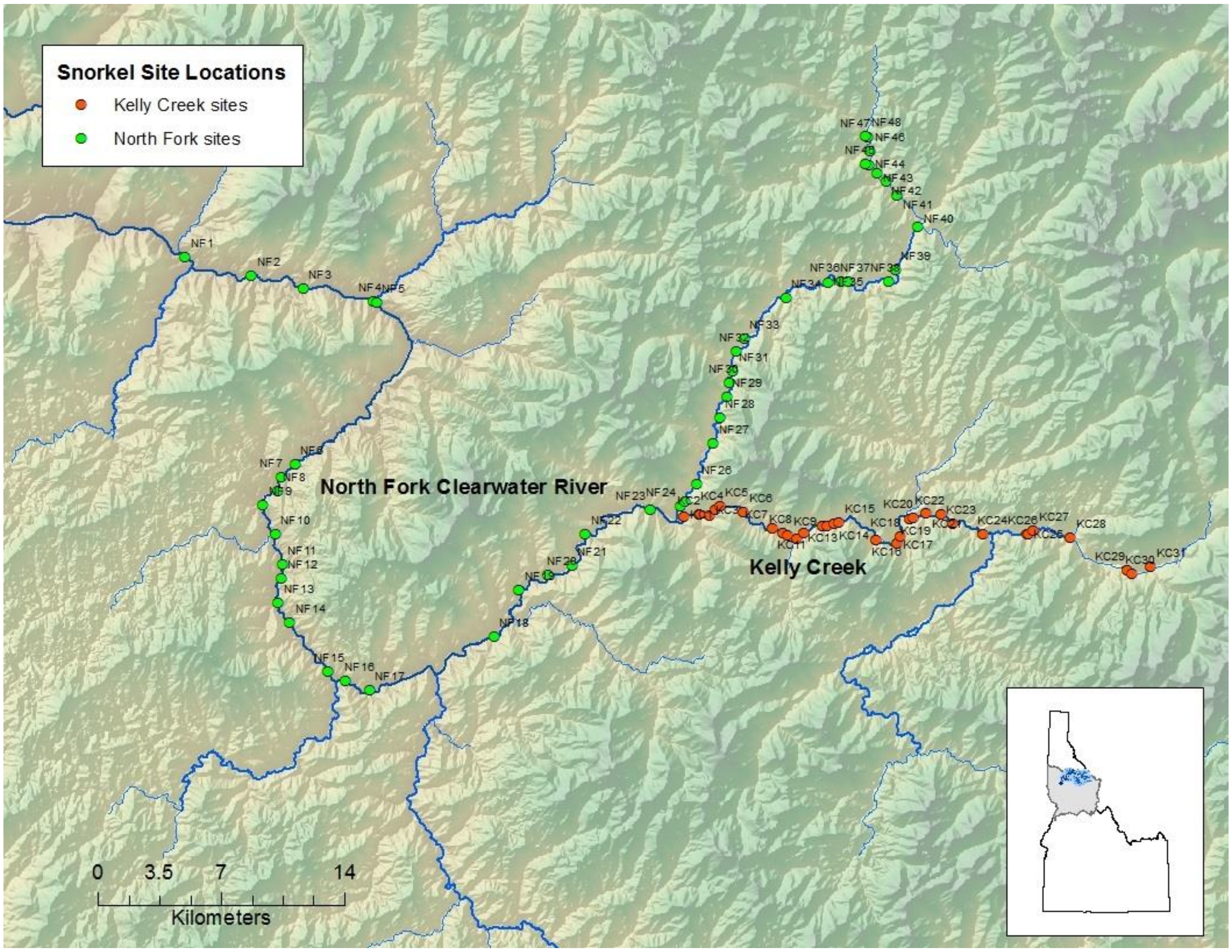


Figure 88. Location of snorkel transects surveyed on the main stem North Fork Clearwater River and Kelly Creek, Idaho, during 2015 and 2016.

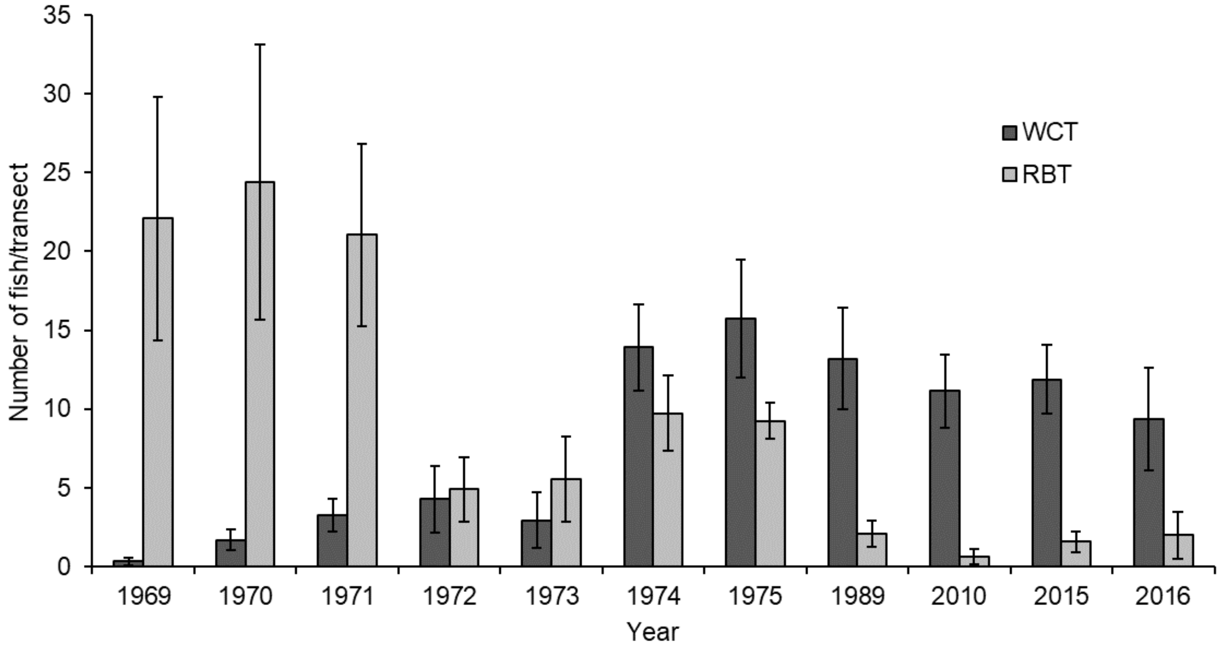


Figure 89. Comparison of number of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) observed per transect for snorkel surveys conducted in Kelly Creek, Idaho, from 1969 to 2016. Error bars represent 90% confidence intervals.

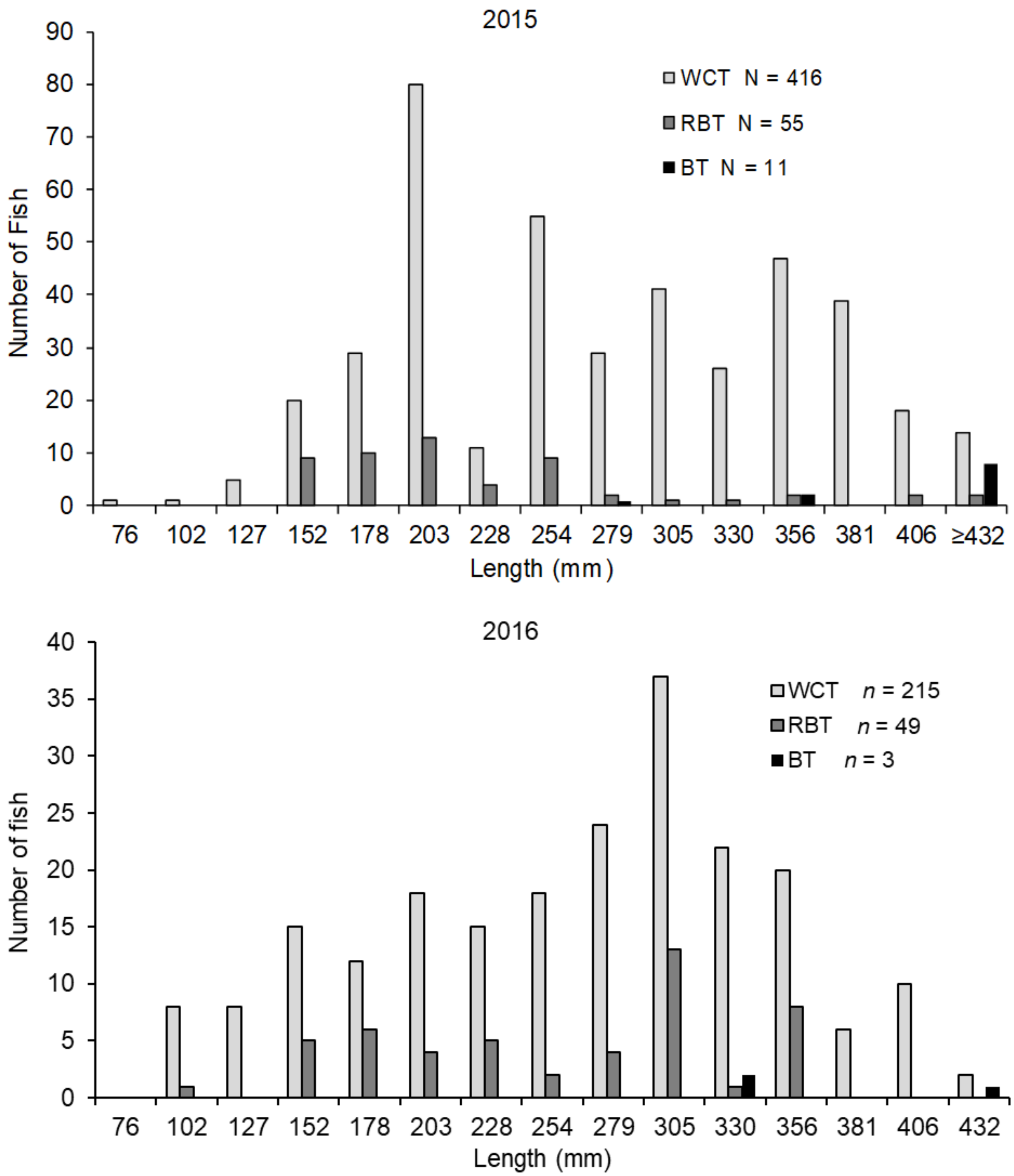


Figure 90. Length-frequency distribution of Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Bull Trout (BT) observed during snorkel surveys conducted in 2015 and 2016 on Kelly Creek, Idaho.

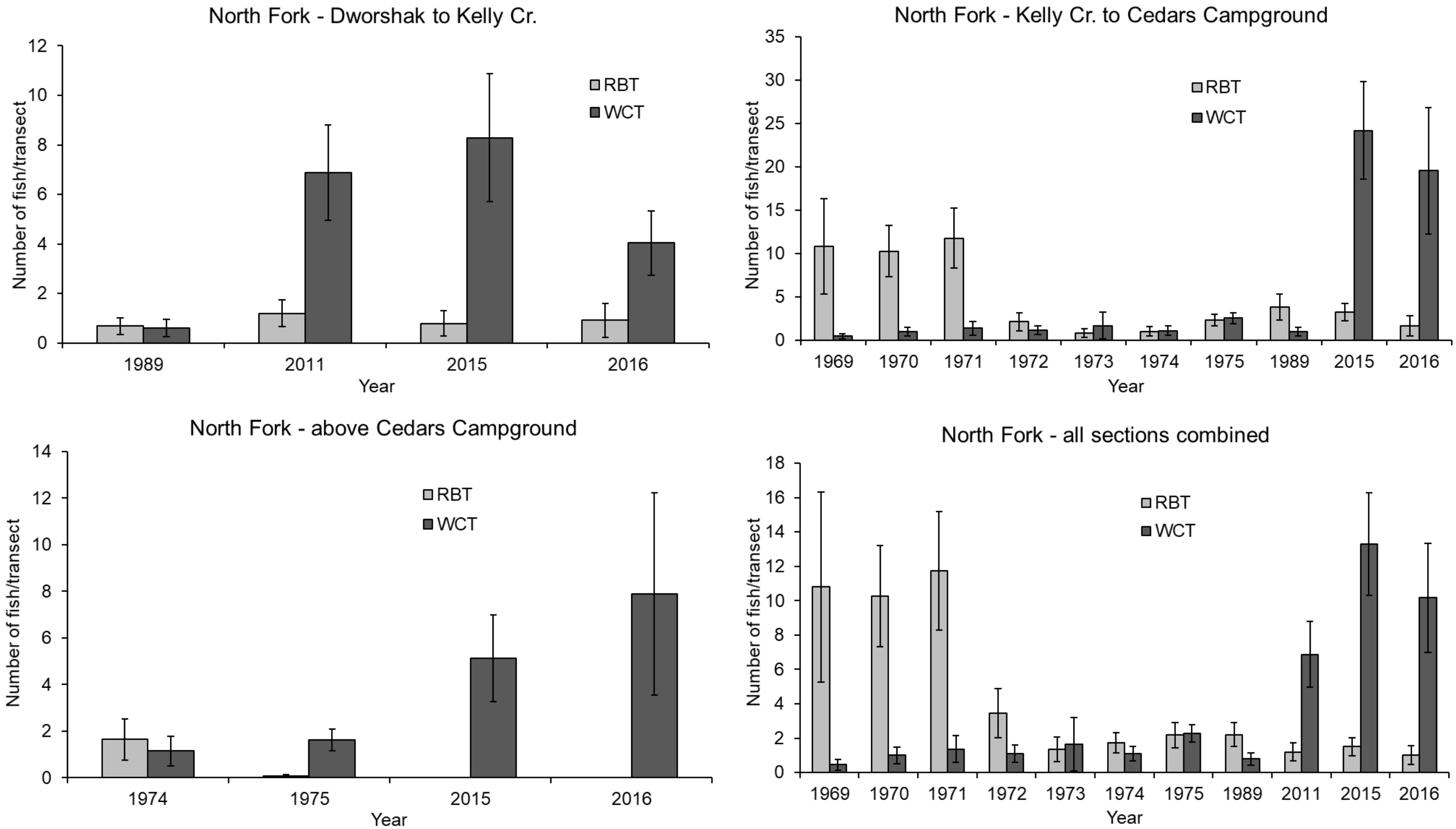


Figure 91. Comparison of number of Westslope Cutthroat Trout (WCT) and Rainbow Trout (RBT) observed per transect for snorkel surveys conducted in the North Fork Clearwater River, Idaho, from 1969 to 2016. Error bars represent 90% confidence intervals.

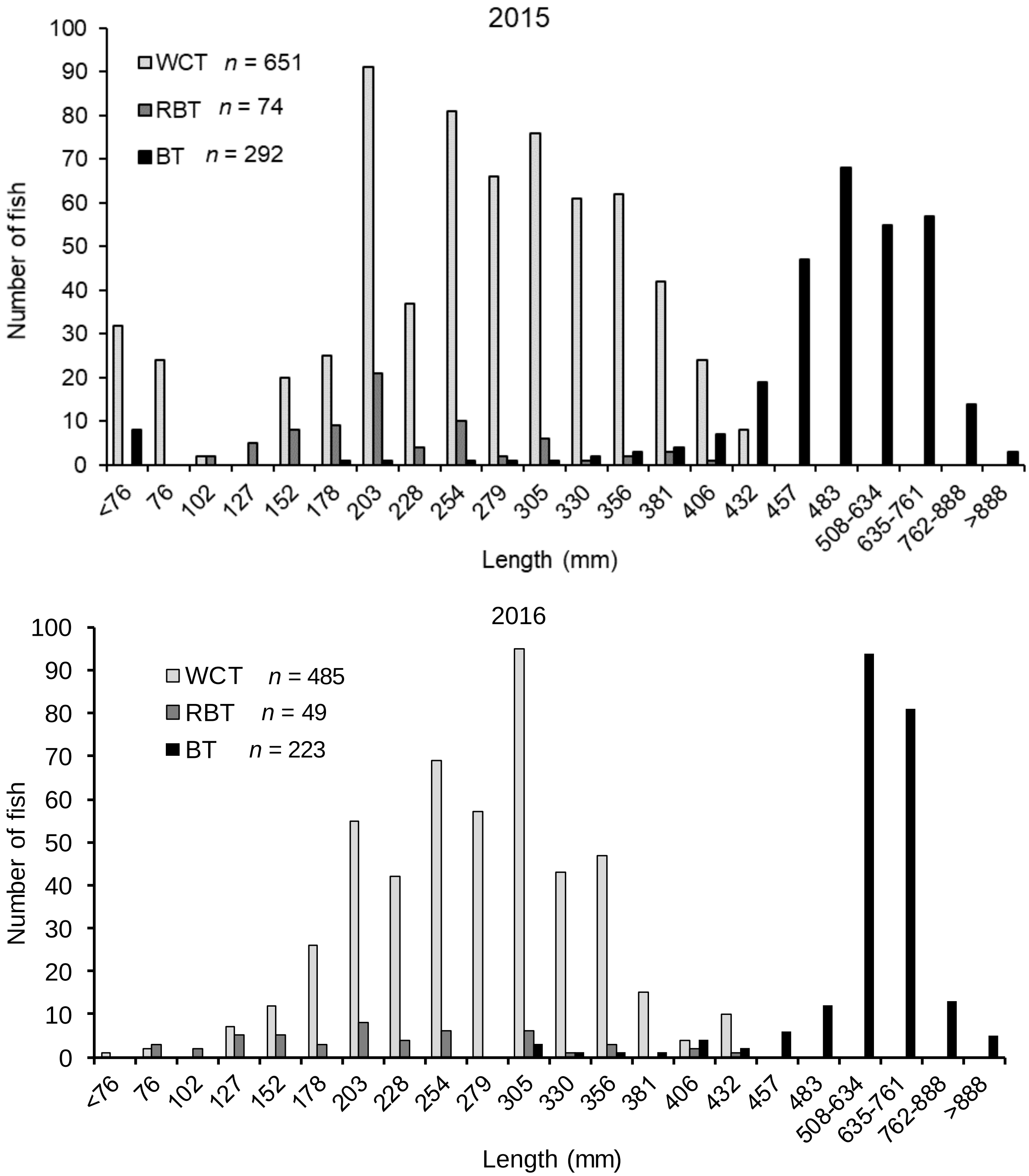


Figure 92. Length-frequency distribution of Westslope Cutthroat Trout (WCT), Rainbow Trout (RBT), and Bull Trout (BT) observed during snorkel surveys conducted in 2015 and 2016 on the North Fork Clearwater River, Idaho.

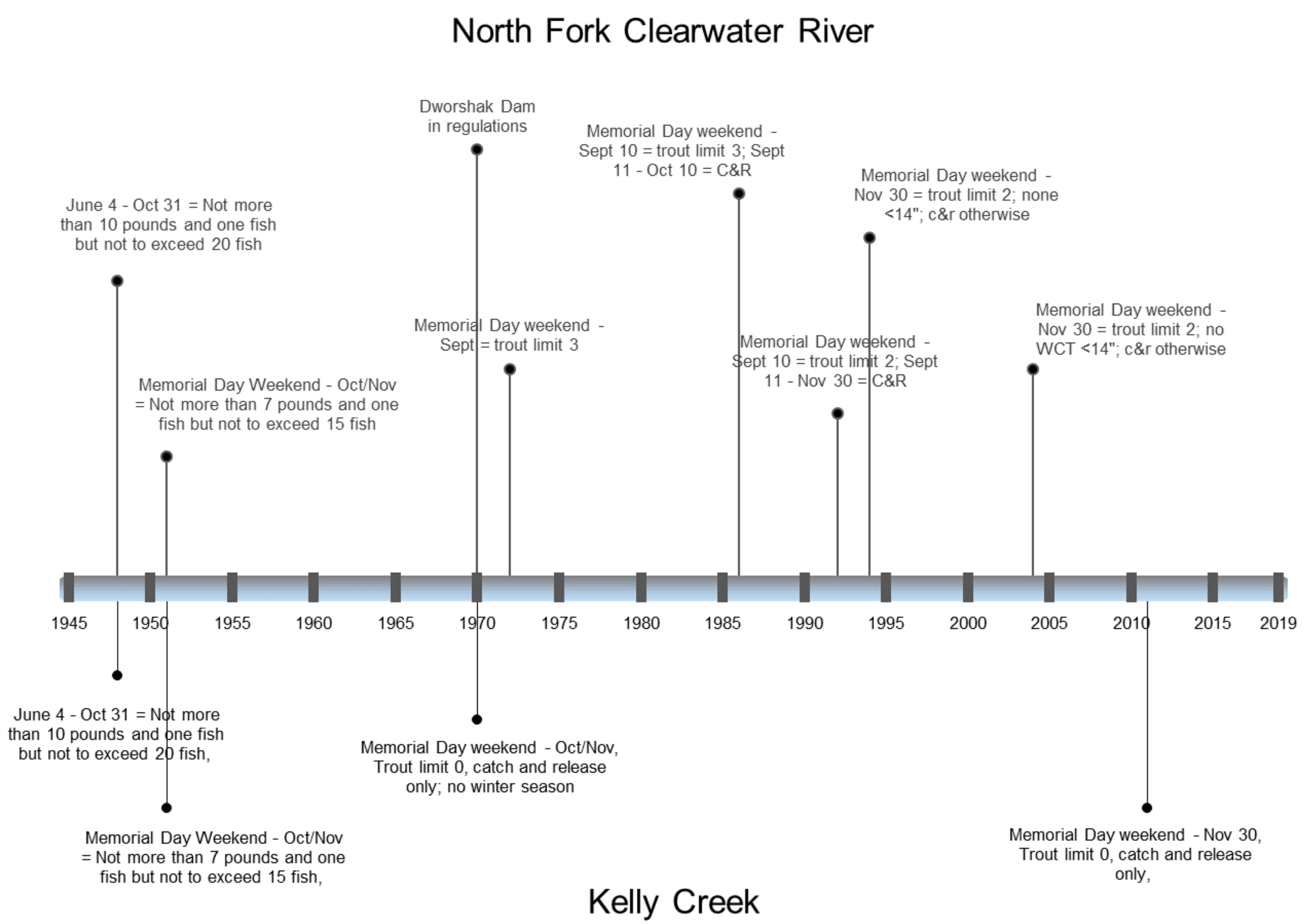


Figure 93. Timeline showing fishing regulations in the North Fork Clearwater River and Kelly Creek, from 1945 to 2019.

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## BULL TROUT REDD SURVEYS


## ABSTRACT

Transects on nine streams were surveyed to enumerate and measure Bull Trout *Salvelinus confluentus* redds, seven of which were index streams for long-term monitoring. A total of 44 redds were identified in the index reaches, which is less than the 15-year mean (mean = 64.1), but slightly higher than the number of redds from the previous year ($n$ = 42). The long-term trend for all index streams was positive, but non-significant ($\tau$ = 0.011, p = 0.999). When analyzed individually, Bostonian Creek exhibited a significant positive trend ($\tau$ = 0.332, $p$ = 0.032) and Lake Creek exhibited a significant negative trend ($\tau$ = -0.397, $p$ = 0.041). Trends for all other streams were non-significant. While counts suggest a recent decline in Bull Trout abundance, abundance does not appear to have decreased below levels observed since 2001 and may be part of normal population cycles. Bull Trout monitoring should be continued so that management actions can be considered if counts continue to decline in the future.



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## INTRODUCTION

Bull Trout *Salvelinus confluentus* are native to Idaho and spawn and rear in many streams and lakes where suitable water temperatures occur (High et al. 2008). In 1998, the US Fish and Wildlife Service listed Bull Trout as threatened under the Endangered Species Act, purporting habitat loss, isolation, non-native fish, and fish passage barriers were threatening their probability of long-term persistence (USFWS 2002). The recovery plan initiated by the USFWS is delineated into six recovery units. The Mid-Upper Columbia Recovery Unit includes the Clearwater River subbasin. The North Fork Clearwater River comprises one of four core areas for Bull Trout recovery within the Clearwater River subbasin (USFWS 2015). Bull Trout in this core area were isolated from other populations in the subbasin by the construction of Dworshak Dam (Hanson et al. 2014), thereby increasing the risk of extinction (Reiman and McIntyre 1996).

Monitoring trends in Bull Trout abundance is an important component in evaluating the status of Bull Trout and the effectiveness of recovery efforts. These fish spawn in the fall when stream flows are low and the water is clear making redd surveys an effective tool in monitoring long-term trends of adult abundance and distribution (Hand et al. 2013). Furthermore, redd counts are available for some tributaries of the North Fork Clearwater from as early as 1994, and for seven index streams as early as 2001, thereby providing a dataset for assessing trends in abundance.

## OBJECTIVE

1. Monitor and assess trends in the spawning population of Bull Trout in the North Fork Clearwater River Core Area.

## STUDY AREA

Redd surveys were conducted in the North Fork Clearwater River drainage, located in the Clearwater National Forest of Idaho (Figure 91). Surveys were conducted on seven index streams, including Bostonian Creek, Placer Creek, Niagra Gulch, Vanderbilt Gulch, Long Creek, Lake Creek, and Goose Creek, all within the upper reaches of the subbasin near the Idaho and Montana border. Additional surveys were conducted on Isabella and Quartz creeks in the lower portion of the subbasin.

## METHODS

The U. S. Forest Service (USFS) conducted redd counts on Bostonian Creek, Placer Creek, Niagra Gulch, and Vanderbilt Gulch. Two passes were conducted by USFS personnel, the first during the week of August 29 and the second during the week of September 12. The Idaho Department of Fish and Game (IDFG) conducted single pass counts on the remaining streams on September 21 and 22. Bull Trout redds were identified based on disturbance to the streambed; depth of the disturbance; color, size, and sorting of the substrate; and stream morphology (Hand et al. 2013). A GPS waypoint was recorded at each identified Bull Trout redd.

Trends in the number of redds were assessed using the annual combined counts for all index streams for years in which all seven were surveyed (Table 22). In addition, each index stream was assessed individually using data from all years a given stream was surveyed. We assessed statistical significance in redd counts trends using Kendall's tau, a nonparametric-rank correlation technique (Reiman and Myers 1997). Spatial distribution of redds among the seven index streams was assessed by calculating the percentage of redds that were counted in a given stream in a given year.

## RESULTS

A total of 44 redds were identified in the seven index streams, which was less than the less than the mean of the past 15-years (mean = 61.4), but slightly higher than 2015 ($n$ = 42). The overall trend for the combined data indicated slow, but non-significant growth ($\tau$ = 0.011, $p$ = 0.999; Table 23). From 2005 to 2010, the number of redds in index streams increased. Since 2010, the number of redds observed declined to the same range observed from 2001 to 2005 (Figure 95).

Individual analysis of the index streams revealed only two with significant trends. Lake Creek had a significant negative trend ($\tau$ = -0.397, $p$ = 0.041; Table 23; Figure 96) and Bostonian Creek was the only index stream exhibiting significant growth ($\tau$ = 0.332, $p$ = 0.032, Table 23; Figure 96). Trends observed for all other index streams were non-significant. A single redd with a pair of fish on it was observed in Quartz Creek and two redds were observed in Isabella Creek. While not a survey stream, a redd was also observed during kokanee surveys in Dog Creek, a tributary to Isabella Creek.

The distribution of redds among index streams has changed over time (Table 24). Prior to 2004, an average of 26% of the redds were counted in Lake Creek. Since then, on average only 6% of the total redds counted in index streams were observed in Lake Creek. Prior to 2003, one of the two lowest redd counts were observed in Bostonian Creek. From 2004 through 2014, one of the two highest redd totals were consistently observed in Bostonian Creek. Since then, very few redds have been observed in Bostonian Creek. Vanderbuilt Gulch produced one of the two highest counts in all but one year on record, but ranged from 28 to 67% of the total count during this period.

## DISCUSSION

Our analysis did not detect a significant trend in the total number of redds for index reaches over the past 16 years. However, the number of redds observed has decreased since 2010, although it has not decreased below numbers observed since 2001. This may be the result of natural fluctuations in the population, rather than signaling a long-term decline. Redd count data from other Bull Trout populations in Northern Idaho and Montana have historically exhibited a considerable degree of inter-annual variation (Reiman and McIntyre 1996, Reiman and Meyers 1997), which suggests that fluctuations in spawning populations may be common. However, monitoring should be continued so that causes and potential management actions could be considered if the decline were to continue.

While trends in redd counts tend to correlate with changes in the abundance of spawning adults, annual variation in redd counts can be confounded by other factors as well (Dunham et al. 2001). These factors include variation in spatial distribution of spawning (Dunham et al. 2001), which we observed in our data. Bull Trout select locations to construct redds based on factors such as stream geomorphology and hyporheic exchange (Baxter and Hauer 2000). Temporal and spatial variation in spawning can change from year-to-year due to annual variation in water temperature and stream flow (Brenkman et al. 2001). Two approaches could be used to help determine the relative importance of changes in distribution driven by environmental variation and actual changes in abundance in relation to redd count trends. First, redd count data could be paired with other indices of abundance that are not to the same biases. Snorkel surveys are already being conducted in the subbasin and may provide such an index, as Bull Trout have been observed in prominent pools staging to spawn. Second, the relationship between environmental factors thought to affect spawning distribution, such as water temperature and streamflow, and redd counts could be assessed to determine the effect of environmental conditions on redd counts.

## MANAGEMENT RECOMMENDATIONS

1. Maintain trend data for Bull Trout redds in tributaries of the North Fork Clearwater River.

2. Investigate the relationship between trends in redd counts and environmental factors that could influence spawning distribution.

3. Investigate independent methods for assessing Bull Trout abundance.

Table 22. Historical data from Bull Trout redd surveys, including the number of redds counted for each stream reach, the number of surveys performed each year, and the number of redds counted in all seven index reaches for years that all seven reaches were surveyed. Index reaches are indicated by grey shading.

| Stream Surveyed | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| North Fork Clearwater River | – | – | – | – | – | – | – | – | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Black Canyon | – | – | – | – | – | – | – | – | 1 | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| *Bostonian Creek* | 0 | 0 | 0 | 0 | 0 | 4 | 1 | 1 | 1 | 18 | 12 | 15 | 14 | 26 | 13 | 15 | 15 | 11 | 4 | 9 | 8 | 2 | 2 |
| Boundary Creek | – | – | – | – | – | – | – | – | – | 2 | 3 | 10 | – | – | – | 0 | – | 12 | – | – | – | – | – |
| Collins Creek | – | – | – | – | – | – | – | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| *Goose Creek* | – | – | – | – | – | – | | 1 | 0 | 2 | 1 | 12 | 8 | 1 | 0 | 2 | 0 | 3 | – | 4 | 8 | 2 | 5 |
| Hidden Creek | – | – | – | – | – | – | – | – | 1 | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Isabella Creek | – | – | – | – | – | – | – | – | 1 | 1 | 0 | 0 | – | 1 | 1 | – | – | – | – | 0 | – | – | 2 |
| Kelley Creek - NFK | – | – | – | – | – | – | – | 14 | – | – | – | – | – | – | – | 6 | – | – | – | – | – | – | – |
| *Lake Creek* | – | – | – | – | – | – | 19 | 7 | 20 | 14 | 5 | 2 | 5 | 3 | 0 | 2 | 0 | 4 | – | 1 | 4 | 5 | 4 |
| Little Moose Creek | – | – | – | – | – | – | – | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| *Long Creek* | – | – | – | – | – | – | 0 | 0 | 5 | 0 | 8 | 10 | 1 | 6 | 10 | 11 | – | 4 | – | 3 | 8 | 2 | 3 |
| *Moose Creek* | – | – | – | – | – | – | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | – | 0 | – | – | – | – | – | – |
| *Niagra Gulch* | – | – | – | – | – | – | 2 | 5 | 6 | 10 | 3 | 4 | 2 | 2 | 2 | 4 | 6 | 2 | 1 | 5 | 4 | 3 | 1 |
| Orogrande Creek | – | – | – | – | – | – | – | – | – | – | – | 0 | – | – | – | – | – | – | – | – | – | – | – |
| Osier Creek | – | – | – | – | – | – | 3 | 0 | 2 | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – |
| *Placer Creek* | 3 | 1 | 2 | 2 | 2 | 7 | 4 | 2 | 4 | 6 | 2 | 3 | 5 | 2 | 3 | 1 | 3 | 1 | 3 | 7 | 2 | 0 | 4 |
| Pollock Creek | – | – | – | – | – | – | – | – | – | 1 | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Quartz Creek | – | – | – | – | – | – | – | 4 | 0 | 0 | 0 | 0 | – | – | 8 | – | – | – | – | 0 | 0 | – | 1 |
| Ruby Creek | – | – | – | – | – | 0 | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Skull Creek | – | – | – | – | – | – | – | – | 0 | 6 | 5 | 3 | – | 4 | 9 | – | – | – | – | – | – | 4 | – |
| Slate Creek | – | – | – | – | – | – | – | – | – | – | – | 3 | – | – | – | 0 | – | 0 | – | – | – | – | – |
| Swamp Creek | – | – | – | – | – | – | 2 | 0 | 1 | 0 | 0 | 2 | – | 1 | – | – | – | – | – | – | – | – | – |
| Upper NF | – | – | – | – | – | – | – | – | – | 7 | 3 | 6 | – | – | – | 0 | – | 14 | – | – | – | – | – |
| *Vanderbilt Gulch* | – | – | – | – | – | – | | 24 | 18 | 13 | 12 | 41 | 35 | 39 | 43 | 49 | 57 | 31 | 33 | 32 | 31 | 28 | 25 |
| Weitas Creek | – | – | – | – | – | – | 1 | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Windy Creek | – | – | – | – | – | 2 | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Breakfast Creek | | | | | | | | | | | | | | | | | | | | | | | |
| Floodwood Creek | – | – | – | – | – | – | – | – | 4 | 0 | 0 | – | – | – | – | – | – | – | – | – | – | – | – |
| Gover Creek | – | – | – | – | – | – | – | – | – | 1 | 0 | – | – | – | – | – | – | – | – | – | – | – | – |
| Stony Creek | – | – | – | – | – | – | – | – | 4 | 0 | 0 | – | – | – | – | – | – | – | – | – | – | – | – |
| Little North Fork Clearwater | | | | | | | | | | | | | | | | | | | | | | | |
| Buck Creek | – | – | – | – | – | – | – | – | – | 5 | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Canyon Creek | – | – | – | – | – | – | – | – | – | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Butte Creek | – | – | – | – | – | – | – | 5 | 0 | – | – | – | – | – | – | – | – | – | – | – | – | – | – |
| Rutledge Creek | – | – | – | – | – | – | – | – | – | 1 | 1 | 6 | 0 | – | – | – | – | – | – | – | – | – | – |
| Rocky Run Creek | – | – | – | – | – | – | – | – | 5 | 1 | 3 | 21 | 13 | 8 | – | 8 | 10 | 1 | 14 | – | – | – | – |
| Lund Creek | 0 | 7 | 2 | 2 | 1 | 1 | 13 | 5 | 7 | 8 | 5 | 19 | 7 | 30 | 22 | 11 | 6 | 1 | 8 | – | – | 3 | – |
| Little Lost Lake Creek | 0 | 1 | 1 | 1 | 7 | 3 | 1 | – | 6 | 7 | 16 | 1 | 38 | 36 | 14 | 5 | 19 | 1 | 2 | – | – | 1 | – |
| Lost Lake Creek | 0 | 0 | 0 | 0 | – | 1 | – | – | 0 | – | 1 | – | 10 | 13 | 8 | 9 | 7 | 6 | 5 | – | – | 1 | – |
| 1268 Bridge to Lund Cr. | – | – | – | – | – | – | – | 17 | 6 | 13 | 8 | 16 | 18 | 20 | 13 | 3 | 6 | 19 | 14 | – | – | 18 | – |
| Lund Cr. to Lost Lake Cr. | – | – | 3 | 1 | 9 | 8 | 3 | 12 | 7 | 7 | 5 | 8 | 16 | 21 | 9 | 11 | 9 | 11 | 16 | – | – | 13 | – |
| Lost Lake Cr. to headwaters | 0 | 2 | 0 | 0 | – | 5 | 1 | – | 5 | 6 | 5 | 11 | 13 | 8 | 20 | 14 | 7 | 6 | 31 | – | – | 1 | – |
| Number of Surveys | 6 | 6 | 7 | 7 | 5 | 9 | 14 | 18 | 26 | 29 | 25 | 23 | 16 | 18 | 17 | 18 | 14 | 17 | 11 | 9 | 8 | 14 | 9 |
| Total Redds for all streams | 3 | 11 | 8 | 6 | 19 | 31 | 50 | 97 | 104 | 129 | 98 | 193 | 185 | 221 | 175 | 151 | 145 | 127 | 131 | 61 | 65 | 83 | 47 |
| Total Redds for 7 index tribs | | | | | | | | 40 | 54 | 63 | 43 | 87 | 70 | 79 | 71 | 84 | | 56 | | 61 | 65 | 42 | 44 |

Table 23. Results of Mann-Kendall trend tests used to analyze redd count data for seven index reaches in the North Fork Clearwater subbasin surveyed by the U.S. Forest Service (USFS) and Idaho Department of Fish and Game (IDFG). The analysis was conducted separately for each stream and for all streams combined. The Kendall's tau ($\tau$) and p value ($p$) are given for each.

| Agency | Stream | $\tau$ | $p$ |
|---|---|---|---|
| USFS | Bostonian Creek | 0.332 | 0.033* |
| | Niagra Gulch | 0.244 | 0.207 |
| | Placer Creek | -0.004 | 0.999 |
| | Vanderbilt Gulch | 0.059 | 0.787 |
| IDFG | Goose Creek | 0.270 | 0.192 |
| | Lake Creek | -0.397 | 0.041* |
| | Long Creek | 0.206 | 0.318 |
| | Combined | 0.011 | 0.999 |

Table 24. The percentage of redds counted in each of seven index streams out of the total that were counted for all index streams during that year.

| Year | Bostonian | Goose | Lake | Long | Niagra | Placer | Vanderbuilt |
|---|---|---|---|---|---|---|---|
| 2001 | 3% | 3% | 18% | 0% | 13% | 5% | 60% |
| 2002 | 2% | 0% | 37% | 9% | 11% | 7% | 33% |
| 2003 | 29% | 3% | 22% | 0% | 16% | 10% | 21% |
| 2004 | 28% | 2% | 12% | 19% | 7% | 5% | 28% |
| 2005 | 17% | 14% | 2% | 11% | 5% | 3% | 47% |
| 2006 | 20% | 11% | 7% | 1% | 3% | 7% | 50% |
| 2007 | 33% | 1% | 4% | 8% | 3% | 3% | 49% |
| 2008 | 18% | 0% | 0% | 14% | 3% | 4% | 61% |
| 2009 | 18% | 2% | 2% | 13% | 5% | 1% | 58% |
| 2011 | 20% | 5% | 7% | 7% | 4% | 2% | 55% |
| 2013 | 15% | 7% | 2% | 5% | 8% | 11% | 52% |
| 2014 | 12% | 12% | 6% | 12% | 6% | 3% | 48% |
| 2015 | 5% | 5% | 12% | 5% | 7% | 0% | 67% |
| 2016 | 5% | 11% | 9% | 7% | 2% | 9% | 57% |

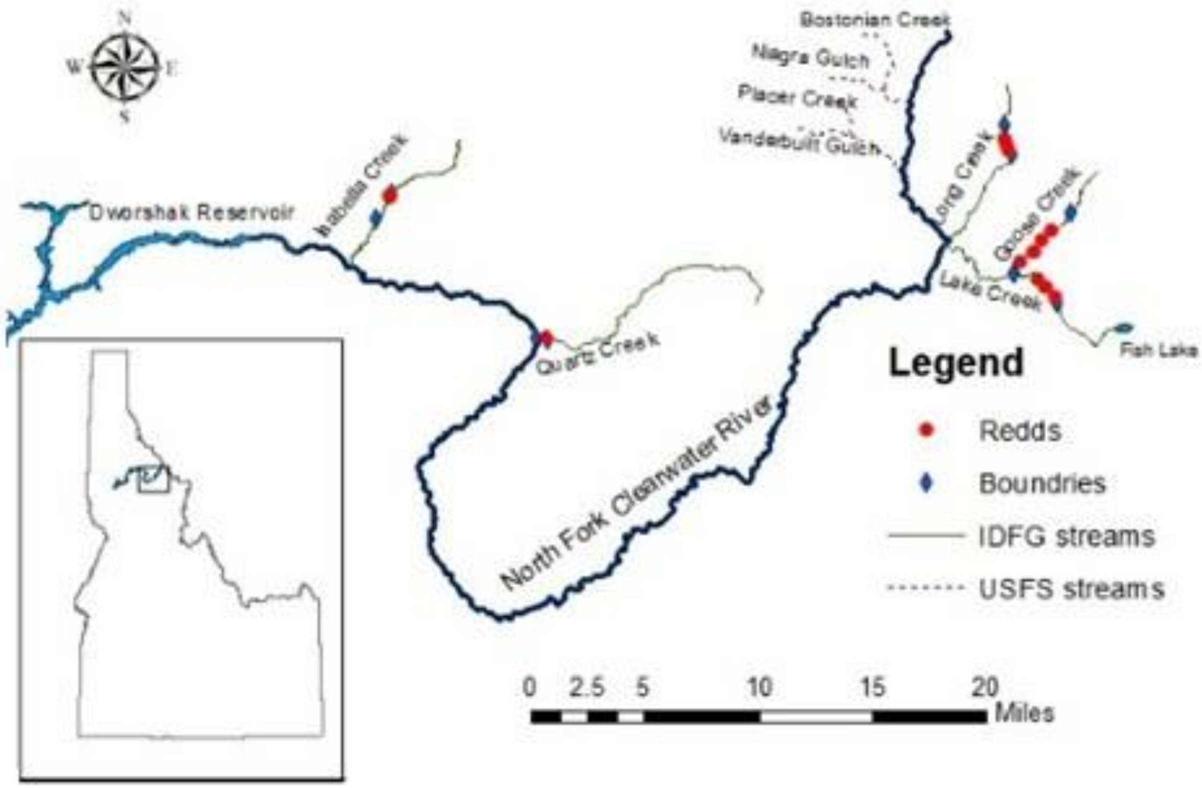


Figure 94. Locations of reaches surveyed for Bull Trout redds in 2013. Streams surveyed by IDFG personnel are indicated by solid lines. Boundaries of the survey reaches are indicated by diamonds and redds identified during the surveys are indicated by circles. Streams surveyed by USFS personnel are indicated by dotted lines, but locations of reach boundaries and redds are not shown.

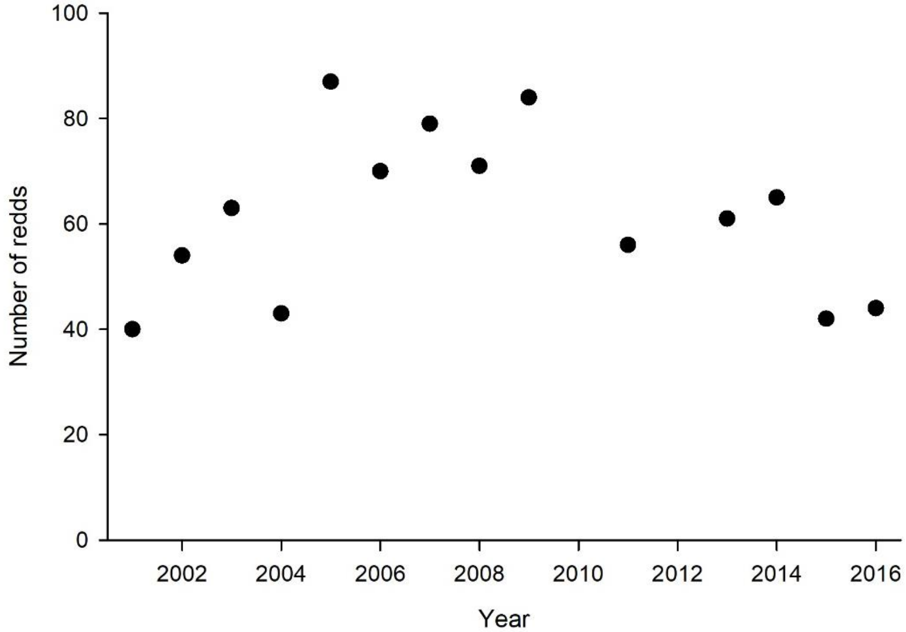


Figure 95. Combined number of Bull Trout redds counted annually in seven index reaches in the North Fork Clearwater River subbasin.

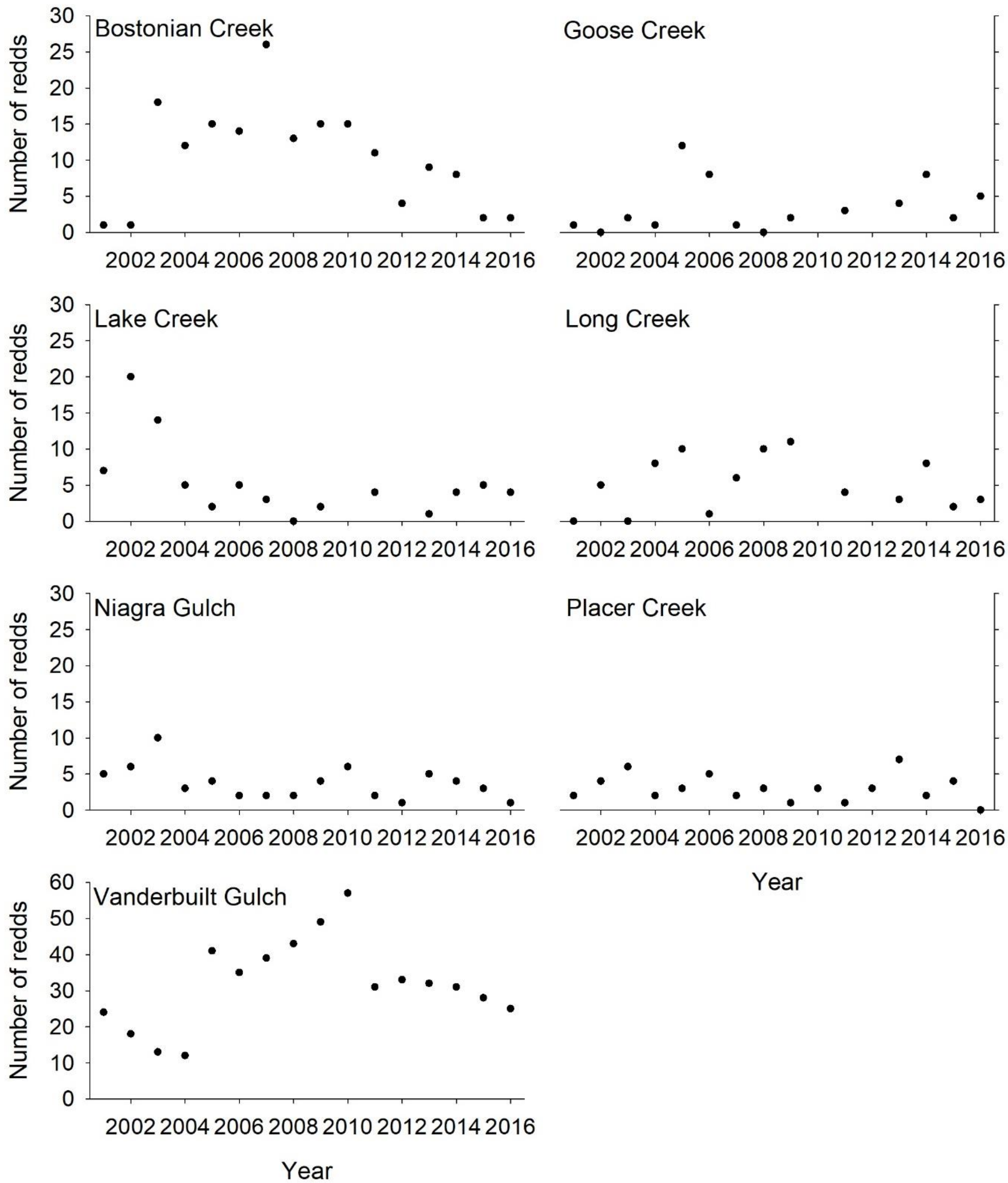


Figure 96. Number of Bull Trout redds counted annually in each of seven index reaches in the North Fork Clearwater River subbasin.

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# EVALUATION OF TIGER MUSKELLUNGE AS A BIOLOGICAL CONTROL AGENT FOR BROOK TROUT IN HIGH MOUNTAIN LAKES


## ABSTRACT

In 2006, through cooperative efforts between the Idaho Department of Fish and Game (IDFG) and the United States Forest Service (USFS), tiger muskellunge (TM; male Northern Pike *Esox lucius* x female Muskellunge *E. masquinongy*) were stocked at high densities (40/ha) into four lakes containing Brook Trout (BKT) *Salvelinus fontinalis* populations. The purpose of this project was to evaluate the potential for TM to eliminate or suppress the BKT populations, decreasing the threat to downstream native fish assemblages. The lakes included were Fly, Heather, and Platinum lakes in the North Fork Clearwater River watershed and Running Lake in the Upper Selway River watershed. In 2016, we resampled Fly, Heather, and Platinum lakes to look at long-term effects of TM on these BKT populations. Our survey resulted in no BKT being captured or observed in Fly and Heather lakes, and one TM captured in Fly Lake. In contrast, Platinum Lake still contained BKT, and although gill net CPUE was similar to previous surveys, anglers caught numerous fish and observed high BKT densities. It appears that TM are no longer present in Platinum Lake. Factors including fish behavior, lake habitat characteristics, and TM mortality all likely contributed to the probable disappearance of these fish. If deemed appropriate, further management actions on these lakes have the potential to eradicate BKT and reduce risk to downstream populations of native fish. Such actions could include additional stockings of TM, intensive gillnetting, rotenone applications, and the use of hatchery YY BKT. At this time, we recommend using eDNA to determine what species are still present in each lake, and then developing a long-term plan for future management of these lakes.



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## INTRODUCTION

Brook Trout (BKT) *Salvelinus fontinalis* were introduced into mountain lake ecosystems throughout Idaho in the early to mid-20th century. Non-native salmonids introduced into mountain lakes often serve as a source population for individuals to enter connecting tributaries, which can result in competitive pressure on native fish (Adams et al. 2001). Additionally, these fish can become stunted in lake environments, resulting in reduced quality of angling opportunities. Brook Trout populations in the mountain lakes of the Clearwater River basin are subject to stunting due to relatively low lake productivity, abundant spawning habitat, early age at maturity, and few predators (Donald et al. 1980; Donald and Alger 1989; Hall 1991; Parker et al. 2001). Where high densities occur, BKT immigration from high elevation lakes to connected streams often increases (Adams et al. 2001). High elevation streams in Idaho contain some of the strongest Westslope Cutthroat Trout (WCT) *Oncorhynchus clarki lewisi* and Bull Trout (BT) *Salvelinus confluentus* populations. Brook Trout can coexist with native fish species (Gunckel et al. 2002), but have been found to reduce or even eliminate BT populations through competitive interaction, predation, and hybridization (Scott and Crossman 1973; Balon 1984; Markle 1992; Rieman and McIntrye 1993). Bull Trout populations within Idaho are currently listed as a threatened species under the Endangered Species Act. Brook Trout have also been found to displace native WCT through interspecific competition (Irving 1987; De Staso and Rahel 1994; Dunham et al. 2002).

In addition to impacting native fishes, stunted BKT populations may also discourage anglers from visiting lakes due to low numbers of quality size (>254 mm) fish present (Rabe 1970; Donald and Alger 1989). Where stunted BKT populations occur, improvements to the fishery can be achieved by reducing their abundance to increase growth rates or by eradicating the BKT so that native salmonids may be introduced. However, the suppression of self-sustaining BKT populations is difficult (Dunham et al. 2002). One technique that has been attempted to eliminate or control BKT in mountain lakes is the introduction of tiger muskellunge (male Northern Pike *Esox lucius* x female Muskellunge *E. masquinongy*; Curet et al. 2008; Koenig et al. 2015). Tiger muskellunge (TM) are a biological control option of interest because stocking at high densities has been found to eliminate or reduce BKT from high mountain lakes in some cases (Curet et al. 2008; Koenig et al. 2015). Work by Koenig et al. (2015) evaluated the response of brook trout in high mountain lakes after stocking high densities of TM (40/ha). The BKT populations were monitored annually for five years after stocking. After five years, BKT populations in the study lakes were eliminated, while others had begun to increase in abundance after initially declining. Since TM have been found to live at least 10 years in mountain lakes (Fitzgerald et al. 1997), we wanted to evaluate longer-term effects of these stockings. To further evaluate the use of TM as a control or suppressing agent for BKT, three of the lakes in this study were surveyed nine years after TM were stocked. These three lakes all occurred in the Clearwater Region and included Fly, Heather, and Platinum lakes.

## OBJECTIVES

1. Evaluate the long-term effects of high density (40 fish/ha) tiger muskellunge introductions on Brook Trout populations in high mountain lakes within the Clearwater Region.

## STUDY AREA

Fly, Heather, and Platinum lakes are located in the in the Five Lake Butte area of the Nez Perce-Clearwater National Forest. They drain into Meadow Creek, a tributary to the upper North

Fork Clearwater River fifth field hydrologic unit code (HUC5) watershed (Figure 97). High densities of Bull Trout and Westslope Cutthroat Trout occur in Meadow Creek (Hanson et al. 2006).

### Fly Lake

Fly Lake is located in a cirque type landform with a north-east aspect and is completely surrounded by sub-alpine forest composed mostly of sub-alpine fir *Abies lasiocarpa* and Engelmann spruce *Picea engelmannii* (Figure 97). Fly Lake is at an elevation of 1,652 m, has a maximum depth of 3.3 m and a surface area of 1.0 ha. Fly Lake's littoral zone is composed mainly of silt. Fly lake has multiple small inlets composed mainly of silt substrates, and a single outlet that drains into a high gradient unnamed stream. Based on the criteria evaluated in Koenig (2015), Fly Lake was thought to have a high potential for successful eradication of Brook Trout due to limited spawning habitat and a migration a short distance downstream of the lakes outlet.

### Heather Lake

Heather Lake is located in a low cirque type landform with a north-west aspect and is surrounded by sub-alpine forest and Idaho granitic bedrock batholithic outcroppings (Figure 97). It is located at an elevation of 1,875 m, has a maximum depth of 9.0 m, and a surface area 2.62 ha. The littoral zone in Heather Lake is composed mainly of silt. There are two inlets to Heather Lake. The major inlet has substrate dominated by silt, sand, and gravel. The main outlet to Heather Lake is located on the southwest side of the lake. The outlet at Heather Lake may be seasonally dry and has a dominant substrate of silt. Heather Lake was thought to have a moderate potential for successful brook trout eradication due to the accessible spawning habitat in the inlet and outlet (Koenig 2015).

### Platinum Lake

Platinum Lake is located in a cirque type landform with a northeast aspect and is surrounded by sub-alpine forest, Idaho granitic bedrock batholithic outcroppings, and talus slopes (Figure 97). It is at an elevation of 1,753 m, has maximum depth of 4.1 m, and a surface area of 1.0 ha. The littoral zone in Platinum Lake is composed of mainly silt. Platinum Lake has multiple small inlets (seeps) and one main outlet that are dominated by rubble and silt. The main inlet and outlet to Platinum Lake are low gradient. Platinum Lake was thought to have a moderate potential for brook trout removal due to the accessible spawning habitat in the outlet (Koenig 2015).

## METHODS

Fly, Platinum, and Heather lakes were sampled on August 15 - 16, 2016 with gill nets and angling. Two floating style monofilament gill nets 36-m long and 1.8-m high were fished in each lake. The nets were composed of six equal size panels with bar mesh sizes of 10.0, 12.5, 18.5, 25.0, 33.0, and 38.0 mm. Monofilament diameter ranged from 0.15 to 0.20 mm. Gill nets were set overnight (19.17 hours), while angling was conducted during the day. Gill nets were set perpendicular to the shoreline in locations that minimized the potential for snagging on underwater obstructions. Angling was conducted using spinners and flies. Capture method, species, total length (mm), and weight (g) was recorded for all fish sampled. We calculated Brook Trout average length and associated ± 90% confidence intervals. Catch-per-unit-effort was also calculated for gill nets (fish/net) and angling (fish/h). Confidence intervals were not calculated for gill nets due to low sample size (two gill nets).

## RESULTS

In Fly Lake, gill net surveys captured no fish, while anglers caught one TM. Thus, the gill net CPUE was 0.0/h, the lowest rate recorded for this lake (Figure 98). Gill net CPUE declined rapidly during the first three years after TM were stocked, and has stayed low since. Several small fish were possibly observed darting under rocks in the lake outlet, although this was not confirmed (they were not observed again). Length-frequency distributions of BKT from previous surveys are shown in Figure 99.

No fish were caught in gill nets or by angling in Heather Lake. This resulted in a gill net CPUE of 0.0/h, the lowest rate recorded for this lake (Figure 98). Gill net CPUE increased slightly the year after stocking TM, but has declined since then. No live fish were observed during sampling in 2016. However, one dead BKT was observed in a shallow bay, indicating that they are still in the reservoir. Length-frequency distributions of BKT from previous surveys are shown in Figure 100.

In Platinum Lake, gill net surveys resulted in the capture of 11 BKT while anglers caught an additional 49 BKT. These fish ranged in length from 125 to 345 mm, with an average of 240 mm (±8; Figure 101). This was the second lowest average length during the study, with sample averages ranging from 203 (±) to 294 (±10) mm. The gill net CPUE was 0.3/h, which was the lowest catch rate of any sample year (Figure 98). Gill net CPUE declined rapidly the year after TM were stocked, then declined slowly through 2016. The angling CPUE was 5.4/h. This was substantially higher than the 0.53 fish/h catch rate for angling in 2011.

## DISCUSSION

In 2011, the initial results of stocking high densities of TM into Fly, Heather, and Platinum lakes indicated these stocking efforts were not successful in eradicating BKT from any of these lakes (Hand et al. 2013). While we observed a large decline in BKT CPUE and increased average length, BKT were sampled in all three lakes five years following TM introduction. In 2016, this trend continued in Platinum Lake, but noticeably changed in Fly Lake and Heather Lake.

In Platinum Lake, the BKT population appears to have reverted back to a high density, stunted BKT population. The large number of fish collected, and the 215 mm average length of BKT caught in gill nets, which was lower than the pre-TM average length of 220 mm in 2006 (Figure 101), confirm that TM were not a successful control agent for BKT in Platinum Lake. This was surprising, as this lake appeared to be a prime candidate for successful eradication based on its habitat characteristics, and the life history of TM and BKT. Tiger muskellunge generally prefer inshore habitat areas with emergent or submerged vegetation and woody debris, as these habitat areas provide better ambush opportunities (Hanson and Margenau 1992; Hand et al. 2013; Koenig et al. 2015). This suggests TM would be more successful in small shallow lakes such as Platinum Lake, as opposed to larger and deeper lakes such as Heather Lake where there is more open water habitat. The continued presence of a large BKT population suggests that something else, such as high mortality of TM is likely a primary cause of failure in this lake. High mortality rates for TM can occur during stocking, with stress-related mortality at stocking as high as 30% (Stein et al. 1981; Carline et al. 1986; Mather et al. 1986). Additionally, survival of age-1 TM stocked in spring has been found to range from 14.3% to 25.3% through the first year (Margenau 1992). Abundance often drops substantially within three to four years (Koenig et al. 2015). The data from our study supports this, as TM catch decreased over a three-year period following stocking and they appeared to be gone after four years (Hand et al. 2013). In contrast to the other lakes in this study, no TM were ever caught or observed in Platinum Lake after they were stocked in 2006 (Hand et al. 2013), suggesting that high mortality of TM occurred quickly in this lake.

In contrast to Platinum Lake, the results of our sampling in 2016 indicate that BKT were either successfully eradicated from Fly Lake and Heather Lake, or reduced to undetectable levels. This data also shows that the time frame for TM to be a successful biological control agent was longer than the original study length of five years. However, we must caution that while no live BKT were caught or observed in these two lakes, there is no guarantee that there are no BKT remaining in the lakes. The presence of a dead BKT in Heather Lake suggests that there are likely a few fish left, just at a very low population level. The capture of one TM in Fly Lake shows how long these fish can survive with little food resources. It has been observed that after BKT had been mostly eliminated, TM stomach contents contained high amounts of dragon fly adults and they were not persisting on BKT alone (Hand et al. 2013). Given the emaciated condition of the TM caught in Fly Lake, it was likely surviving primarily on food other than fish.

The success of BKT suppression in Heather Lake is somewhat surprising. Heather Lake has a maximum depth of nine meters and has a much larger pelagic zone than either Platinum or Fly lakes. Additionally, a large headwall along one side of the lake substantially reduces the quantity of the lake's littoral zone. Heather Lake also has long inlet and outlet streams that provide refuge and spawning habitat. These habitat characteristics were previously thought to be potential hindrances to the success of TM in controlling BKT (Hand et al. 2013). Based on the data collected in 2016, this does not appear to be the case.

Before we consider any plan to restock these lakes, additional sampling and fish removal efforts should be conducted to ensure that no fish remain. Failure to remove all BKT could result in the reestablishment of a BKT population. This would likely inhibit the potential for a successful WCT introduction. In lakes where BKT can escape capture because of inlets, outlets or other habitat features, other removal techniques will be necessary to totally eliminate BKT from a lake. Because BKT catch rates declined post TM introduction, and smaller size classes are evident in Platinum Lake, successful eradication of BKT from this lake may be possible if it is restocked with TM combined with extra removal efforts such as inlet/outlet rotenone, electrofishing treatments, gillnetting, and angling. Sampling techniques, such as using eDNA, should be considered to provide additional information regarding the presence of either species after eradication techniques are implemented.

Although there are many fish removal techniques available, an additional option for removing BKT may be available in the future. Hatchery-raised YY BKT could be used to effectively crash the population after several generations (Schill et al. 2016). This technique appears to have a high potential for success; however, we must ensure that TM are completely removed from any lake stocked with these fish, or they will become prey for TM that remain. Sampling with eDNA, or waiting long enough post-stocking, would be necessary to ensure that no TM remain.

## MANAGEMENT RECOMMENDATIONS

1. Utilize eDNA to determine if Brook Trout were successfully eradicated from Fly and Heather lakes.

2. Utilize eDNA to determine if tiger muskellunge are still present in Heather and Platinum lakes.

3. Explore combining YY Brook Trout with conventional suppression techniques to eradicate Brook Trout from Platinum Lake.

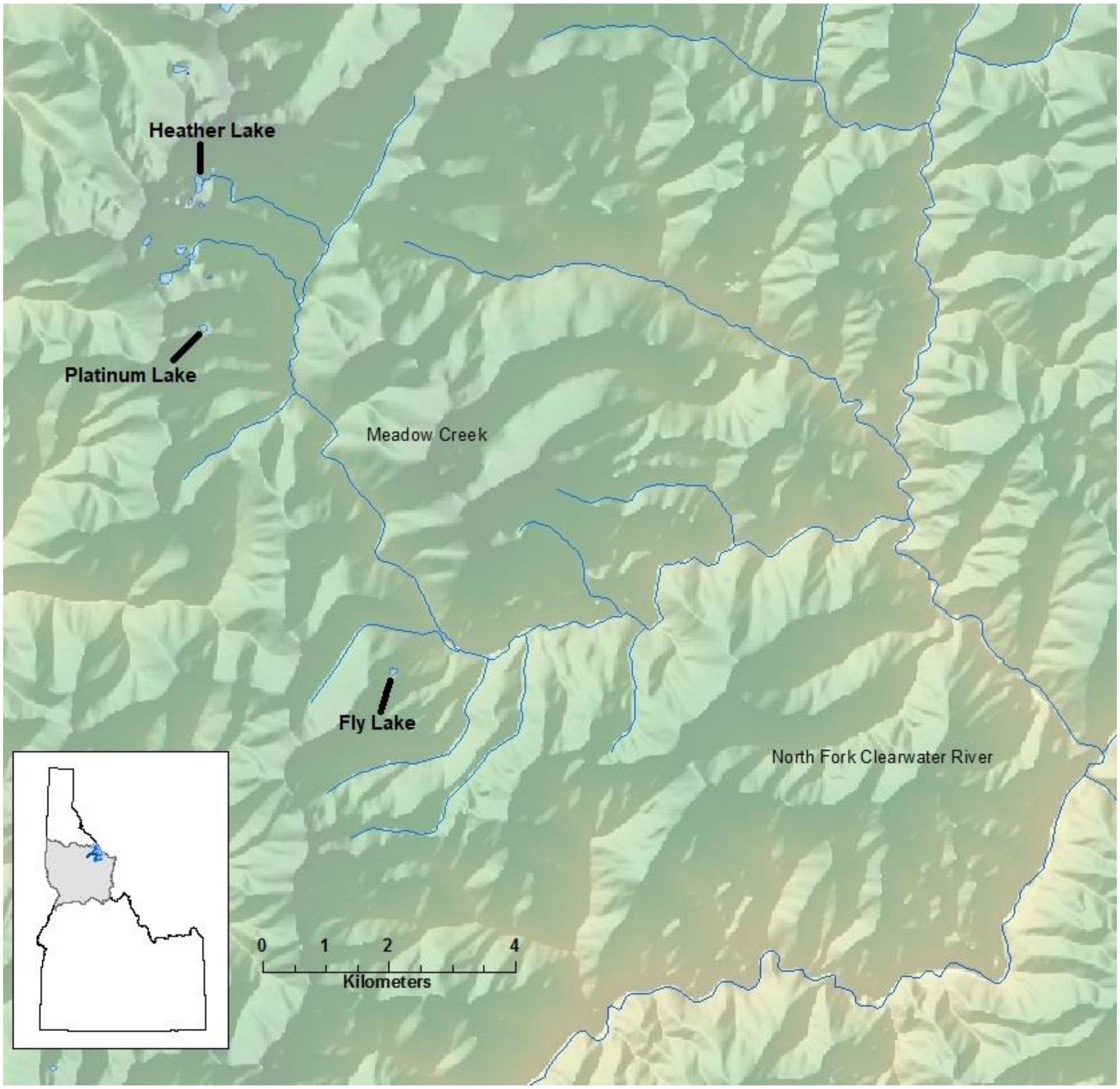


Figure 97. Map showing the locations of Fly, Platinum, and Heather lakes, Idaho.

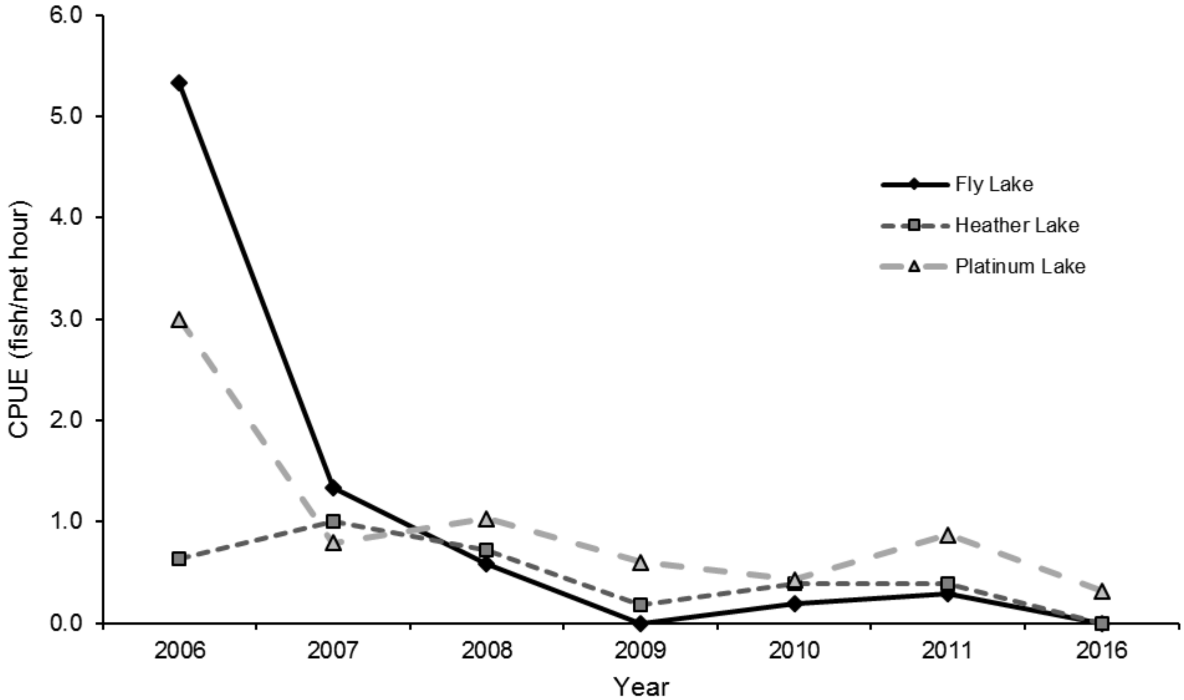


Figure 98. Catch-per-unit-effort (fish/h) of Brook Trout collected by gill nets in Fly, Heather, and Platinum lakes, Idaho, from 2006 to 2016.

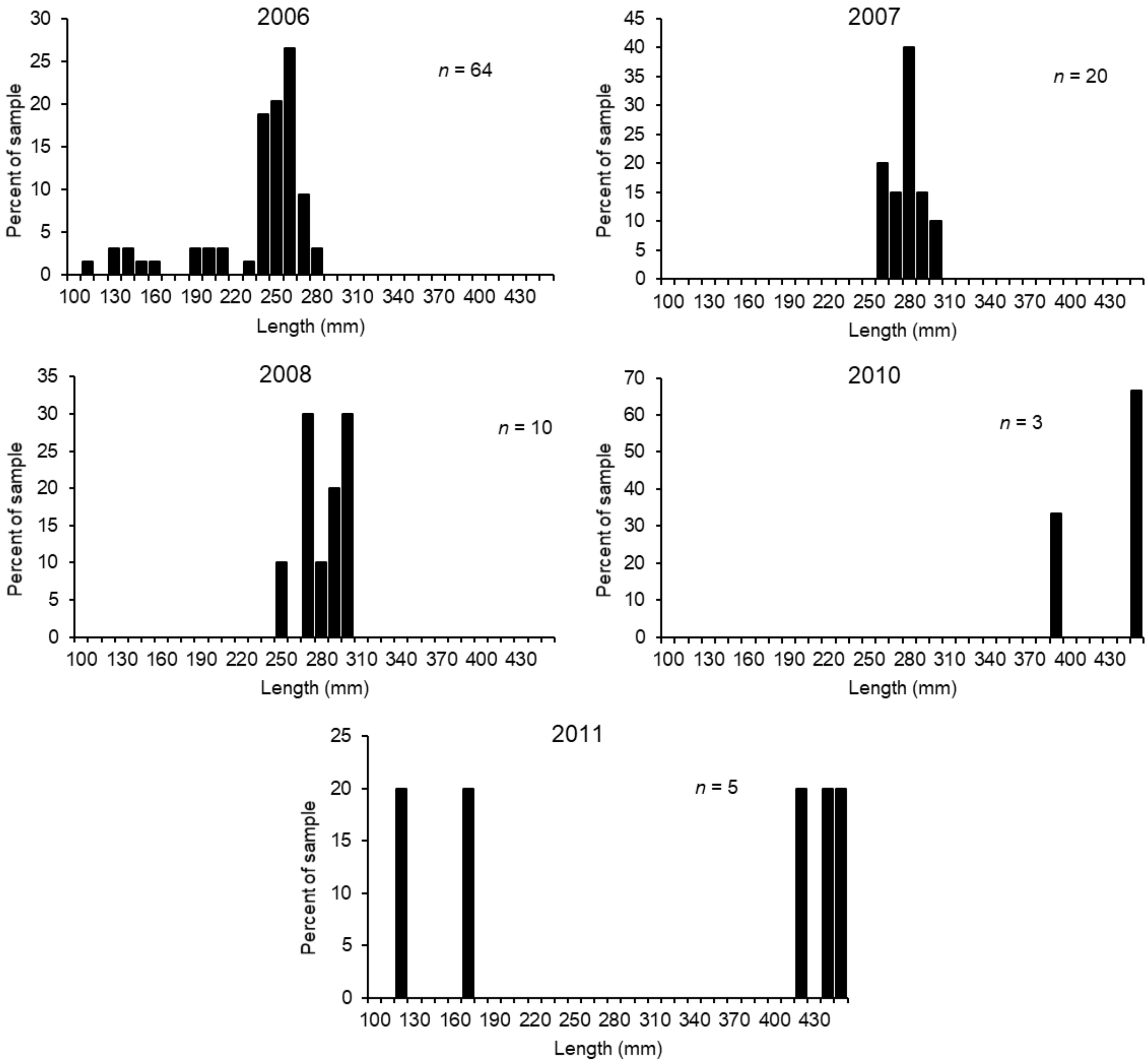


Figure 99. Length-frequency distributions of Brook Trout collected by gill nets in Fly Lake, Idaho, from 2006 to 2016.

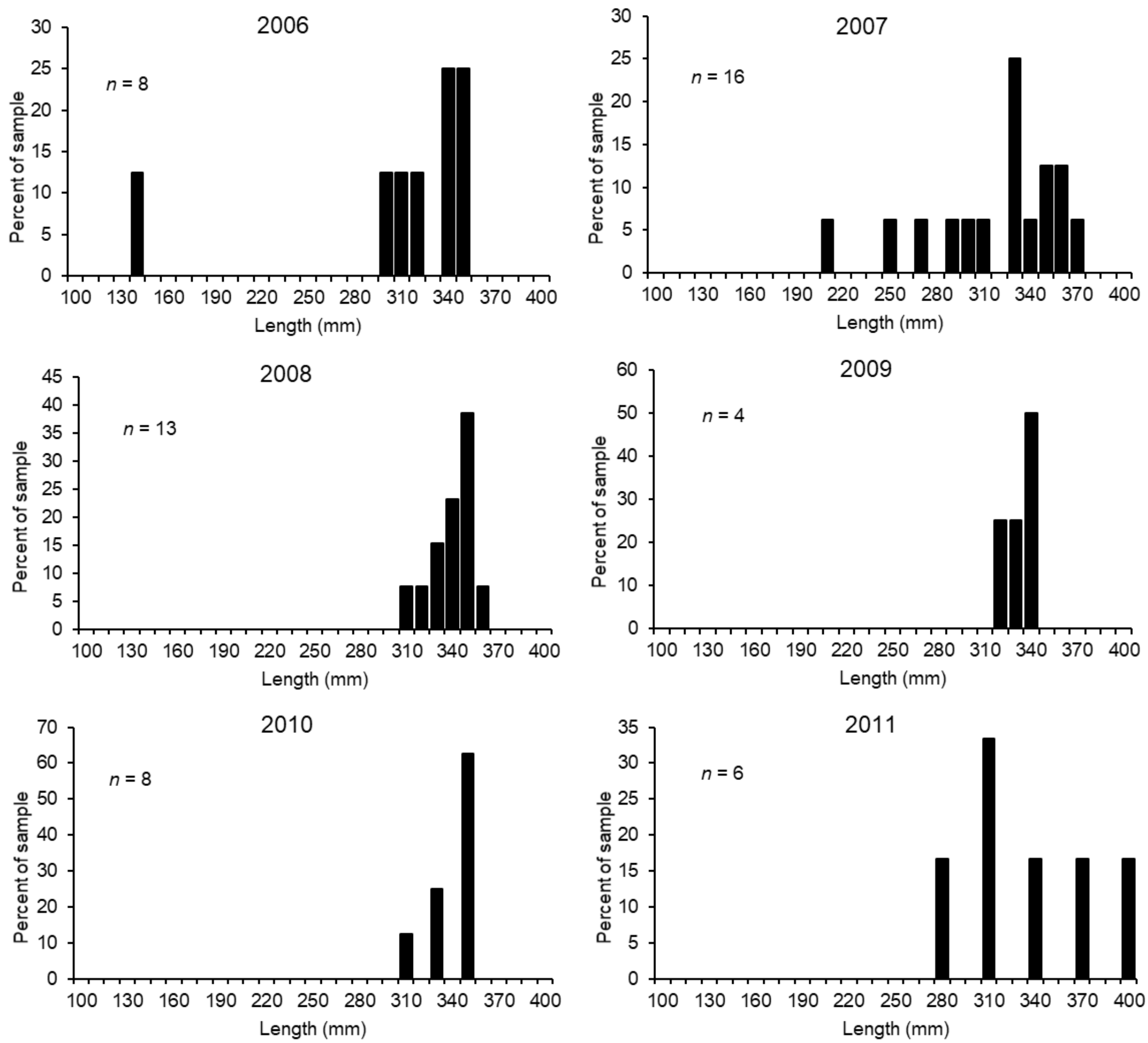


Figure 100. Length-frequency distributions of Brook Trout collected by gill nets in Heather Lake, Idaho, from 2006 to 2016.

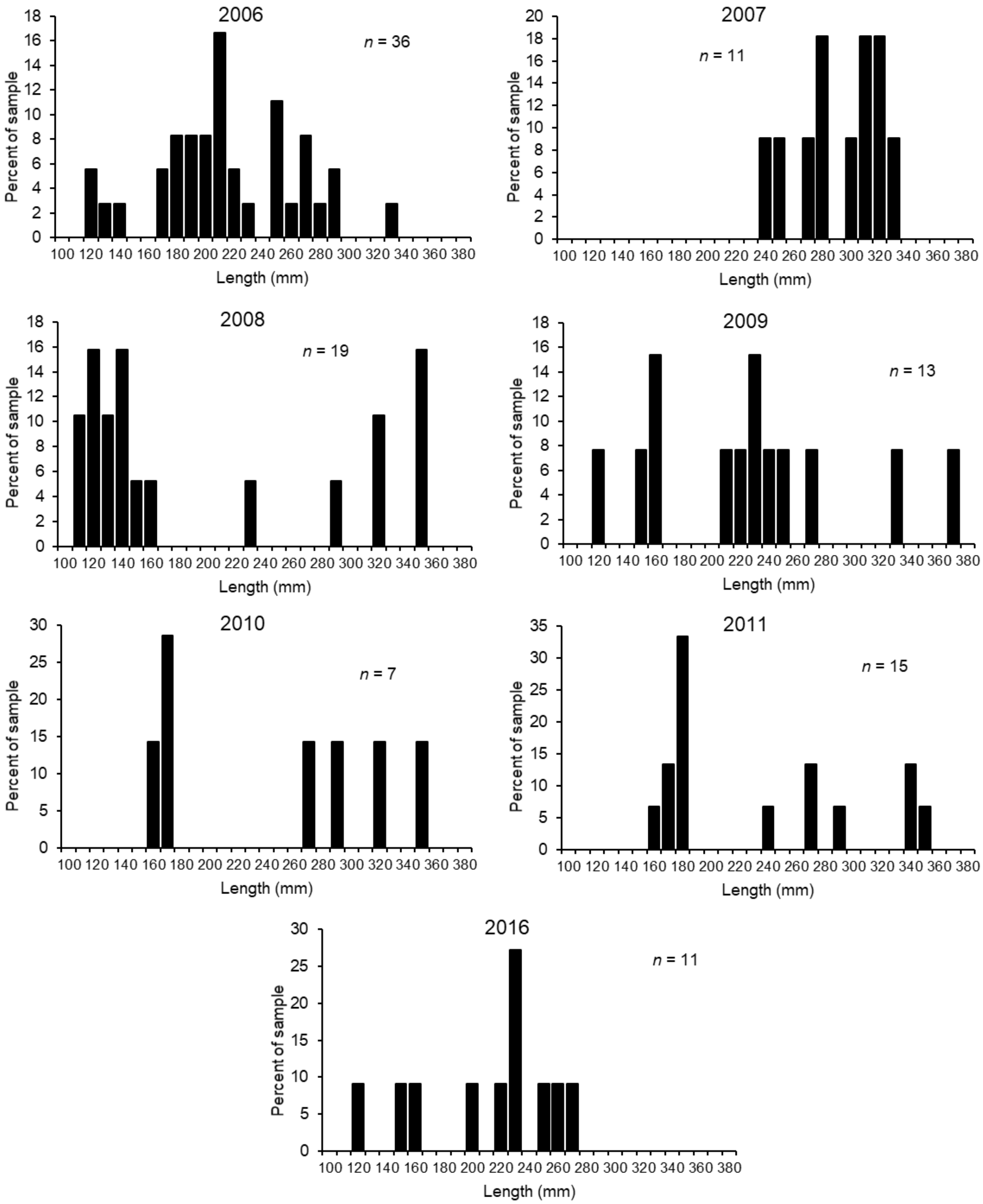


Figure 101. Length-frequency distributions of Brook Trout collected by gill nets in Platinum Lake, Idaho, from 2006 to 2016.

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## MOUNTAIN LAKES MONITORING IN CONSIDERATION OF AMPHIBIAN RISK ASSESSMENT IN NORTH CENTRAL IDAHO

### ABSTRACT

In 2016, we conducted our eleventh year of a 20-year study evaluating long-term trends in amphibian populations within high mountain lakes, and to determine the extent fish stocking is a threat to their persistence. Multiple amphibian visual encounter surveys (VES) were completed on 24 lakes. All 74 lakes included in this study have now been sampled at least twice. In the both the first and second rounds of sampling, 63 of 74 lakes (85.1%) had Columbia Spotted Frogs (CSF) *Rana luteiventris*; however, several lakes went from having frogs present in the first round to absent in the second round and vice-versa. The detectable population of Long-toed Salamanders (LTS) *Ambystoma macrodactylum* changed drastically between the two rounds of sampling. Having been found in 27 of 74 lakes (36.5%) in round one, they were found in 37 of 74 (50.0%) lakes in round two. Long-toed Salamanders were also only found in four lakes that contained fish in the first round and seven in the second. Habitat relationships for both CSF and LTS were generally consistent with previous years' analyses. For CSF, counts of >2 adults and lake depth were positively correlated with occurrence, while the proportion of fine substrate(s) in the lake was positively correlated with both occurrence and abundance. The depth of the spring snowpack was also positively correlated with abundance, while the presence of fish was negative. Long-toed Salamander occurrence and abundance were both negatively correlated with fish presence, while abundance was positively associated with lake depth, spring snowpack depth, and the proportion of fine lake substrate(s). As with previous years' analyses, we found no significant long-term trends in amphibian population occurrence, but we did detect significant positive trends in abundance. However, these positive abundance trends may be due to multiple biases, including surveyor bias, so results must be viewed with caution until more data is collected.

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## INTRODUCTION

Amphibian population reduction and species extinction has given urgency to amphibian conservation, inventory efforts to determine baseline data, and monitoring to determine trends in amphibian populations (Houlahan et al. 2000; Stuart et al. 2004; Beebee and Griffiths 2005; Orizaola and Brãna 2006). Potential factors in amphibian population decline are numerous and include: habitat modification/fragmentation, introduction of predators/competitors, increased UV-B radiation, changes in precipitation/snowpack, and pathogen infection (Alford and Richards 1999; Corn 2000; Marsh and Trenham 2001; Pilliod and Peterson 2001). Throughout the north-central mountains of Idaho, direct (predation) and indirect (resource competition, habitat exclusion, and population fragmentation) impacts on amphibian populations from introductions of trout into historically fishless lakes are a cause for concern (Petranka 1983; Semlitsch 1988; Figiel and Semlitsch 1990; Bradford et al. 1993; Brãna et al. 1996; Tyler et al. 1998). Trout have been stocked into high mountain lakes to provide angling opportunities to backcountry visitors. As much as 95% of previously and/or currently stocked high mountain lakes throughout the western United States that were once fishless, now contain fish through regular stocking efforts or self-sustaining populations from legacy stocking efforts (Bahls 1992). It is estimated that 96% of lakes within the Clearwater National Forest were historically fishless, as the headwater-area topography where lakes are located is relatively steep (Murphy 2002). According to historical stocking records, some lakes in north-central Idaho were stocked as early as the 1930s (Murphy 2002). Out of the estimated 3,000 mountain lakes in Idaho, approximately 1,355 lakes (45%) are stocked or have naturally-reproducing fish populations (IDFG 2012).

Mountain lake ecosystems in North Central Idaho contain amphibians such as Long-toed Salamanders (LTS) *Ambystoma macrodactylum* and Columbia Spotted Frogs (CSF) *Rana luteiventris*, although Idaho giant salamanders (IGS) *Dicamptodon aterrimus*, western toads *Bufo boreas*, and Rocky Mountain tailed frogs *Ascaphus montanus* may also be present. Common reptiles found at these mountain lakes may also include common garter snakes *Thamnophis sirtalis* and western terrestrial garter snakes *T. elegans* (GS), both of which were historically the main predators of amphibians (Murphy 2002). The Idaho Department of Fish and Game (IDFG) Clearwater Region contains 711 mountain lakes. Approximately 400 of these mountain lakes were previously inventoried in the Clearwater Region through cooperation between the IDFG and United States Forest Service (USFS).

Previous studies found that CSF presence (and breeding occurrence) in this area was not significantly different in lakes with or without fish after accounting for habitat effects (CSF were positively associated with increasing amounts of sedge meadow perimeter and silt/organic substrate; Murphy 2002). However, CSF abundance at all life stages was significantly lower in lakes with fish than without fish (Murphy 2002). Long-toed Salamander larvae and/or breeding adult presence and abundance (adults are typically terrestrial except during breeding) was significantly less common in lakes with fish then lakes without fish (Murphy 2002). However, where native (non-stocked) Westslope Cutthroat Trout (WCT) *Oncorhynchus clarkii lewisi* existed in lakes, the impact on LTS was not as severe as compared to lakes that were historically fishless and later stocked with introduced western trout (Murphy 2002). Other studies have examined relationships between introduced trout and salamanders. Direct negative impacts by fish on amphibian populations have been mostly attributed to trout preying upon amphibians when they are in the larval stage, although trout may also cause salamanders to avoid lakes previously used as breeding transects resulting in indirect impacts as well (Kats et al. 1993; Figiel and Semlitsch 1990; Bradford et al. 1993; Knapp 1996; Pilliod et al. 1996; Graham and Powell 1999).

Introduced fish populations may also indirectly impact amphibian gene flow, recolonization, and subsequent persistence. The degree of gene flow in mountain lake amphibian populations likely relies on connectivity between higher and lower elevation subpopulations (with low gene flow). Gene flow may also occur between neighboring lakes that are not necessarily within the same wet stream migration corridor when overland dispersal is not drastically limited by headwater topography, precipitation, and/or canopy cover (Murphy 2002). Long-toed Salamanders within north-central Idaho are panmictic (randomly-interbreeding populations) with high levels of variation within populations providing evidence that populations are not evolving in complete isolation (Tallmon et al. 2000). Amphibian populations, or demes, in these headwater areas likely never evolved with native fish and may lack the appropriate defensive, behavioral, or chemical responses to coexist with introduced fish populations (Kats et al. 1988).

Westslope Cutthroat Trout, Rainbow Trout (RBT) *O. mykiss*, RBT x WCT hybrids, and Brook Trout (BKT) *Salvelinus fontinalis* are the most common introduced fish species in high mountain lakes in the Clearwater Region. Additionally, some lakes within the study area have a stocking history that may include Yellowstone Cutthroat Trout *O. bouvieri*, California Golden Trout *O. mykiss aguabonita* (last stocked in 1990 in the Clearwater Region - Steep Lakes), Arctic Grayling *Thymallus arcticus* (last stocked in 1982 in the Clearwater Region - Bald Mountain Lake), and various forms of trout hybrids. The term “introduced western trout” may be more appropriate for *Oncorhynchus* species in these lakes where natural reproduction is occurring, as the degree of hybridization is unknown in lakes where multiple species have been stocked (Behnke 1992). The Clearwater Region currently stocks 87 of its 711 high mountain lakes. Most lakes are stocked with fingerling WCT on a three-year rotation by fixed wing aircraft.

Certain species of introduced trout tend to have a greater impact on amphibian occupancy than others (Murphy 2002). Brook Trout tend to impact CSF and especially LTS presence and breeding to a greater extent than the presence of either *Oncorhynchus* species. This impact is derived from differences in fish spawning times/behavior and variations in amphibian habitat usage just after ice-off conditions in mountain lakes (Murphy 2002). Westslope Cutthroat Trout and RBT in these lakes spawn in spring/summer which often coincides with times that amphibian breeding occurs. As a result, both fish species are typically preoccupied with spawning in inlets or outlets while amphibians are typically breeding within the lake itself. This difference in spawning habitat use may allow amphibians to breed with fewer disturbances by WCT and RBT (Murphy 2002). In contrast, BKT spawn in the fall and are actively moving and foraging throughout the lake in spring and are more likely to prey upon any amphibian life stage and/or harass breeding adults (Murphy 2002). Furthermore, BKT tend to be more benthic oriented (where salamanders usually occur), seek out larger prey items, and attain higher densities within mountain lakes than *Oncorhynchus* species (Griffith 1974). Columbia spotted frogs do not tend to be impacted by BKT presence to the same magnitude as LTS because of their different habitat associations and shorter larval stage.

Long-toed Salamanders occupy a wide range over the western United States and Canada. The majority of LTS in Idaho sub-alpine lakes have a two-year larval stage, making them susceptible to predation by fish for a longer period of time. Studies suggest that they are more susceptible to impacts by introduced fish than the CSF (Murphy 2002). However, conclusive evidence of LTS decline is insufficient (Graham and Powell 1999). For this reason, a long-term monitoring project (20 years) was initiated in the Clearwater Region to provide knowledge of the amphibian population dynamics within the north-central mountains of Idaho. Long-term monitoring of mountain lakes will allow for amphibian population trends to be identified and will give managers the ability to determine whether sufficient fishless habitat exists to support amphibian populations into the future.

Prior to the 2006 mountain lakes field season, a long-term monitoring study design and protocol was developed for mountain lakes. The study design and protocol addressed the amphibian risk assessment that has been developed through previous studies and inventories of mountain lakes conducted within north-central Idaho (IDFG, *unpublished data*).

The amphibian risk assessment is based on the amount of fishless habitat that exists within a watershed at the HUC5 level. At the individual HUC5 watershed level, it is assumed monitoring will be able to examine conditions that may dictate local response in the interactions of stocked fish and native amphibian populations to provide a more defined opportunity for prioritized management action (Murphy 2002). While there are many risk factors associated with amphibian declines, our assessment focused on considering impacts that may be associated with native and stocked fish in lakes on a HUC5 watershed basis. The amphibian risk assessment for these high mountain lake ecosystems has four categories: control (no risk), low, moderate, and elevated.

- *Control (no risk)* – watershed has never experienced fish introductions through stocking activities.
- *Low* – At least 50% of the lakes within a watershed are fishless <u>AND</u> a minimum 20% of the lake surface area within the watershed is fishless.
- *Moderate* – 50% of lakes within a watershed are fishless <u>OR</u> 20% of surface area is fishless.
- *Elevated* – Meets neither requirement, less than 50% of the lakes within a watershed are fishless <u>AND</u> less than 20% of the surface area within the watershed is considered fishless.

Two watersheds (HUC5) were selected randomly from each of the amphibian risk categories (region-wide from all HUC5 watersheds that contained lakes) for sampling. This resulted in eight HUC5 watersheds containing 72 lakes within the Nez Perce-Clearwater National Forest. In 2013, a third randomly selected control watershed (Big Harrington Creek in the Bitterroot National Forest was added to increase the sample size of fishless control lakes, bringing the study's total to nine watersheds that contain 74 lakes. Attempts will be made to sample all lakes within a selected HUC5 watershed within the same field season. The 20-year period for the high mountain lakes long-term monitoring project will allow for each of these lakes be sampled five different times. The repetition of sampling events will allow for comparisons to be made within (for trends) and between watersheds (for comparisons among amphibian risk classes). In addition, repetition of sampling events will address the normal patterns of recruitment fluctuations often common among amphibian populations. Sampling frequency and rotation order are adjusted to accommodate weather and fire conditions.

## OBJECTIVES

1. Evaluate the relationships of fish and amphibians within high mountain lake ecosystems in the IDFG Clearwater Region.

2. Assess whether current fisheries management strategies in high mountain lakes of North Central Idaho adequately balance recreational fishing opportunity and provide for the long-term persistence of amphibian populations.

## STUDY AREA

The 74 lakes selected for this study are located in the Bitterroot National Forest and the Nez Perce-Clearwater National Forest, both located in north-central Idaho (Figure 102). In 2016, IDFG personnel surveyed 24 lakes within four HUC5 watersheds: Old Man Creek and Storm Creek in the Clearwater National Forest and Bargamin and Running Creek in the Nez Perce National Forest (Figure 102). Photographs, routes and bathymetric/surrounding area maps of lakes within the HUC5 watersheds are maintained in the Clearwater Region office within the mountain lakes database. As of 2016, not all of these files are complete, and will require completion in following years of the study. Available files are located in the IDFG Clearwater Region shared drive at the address: S:\Fishery\MTN Lakes\Long Term Monitoring\Photos, Lake Maps, Routes.

## METHODS

Field sampling was conducted following the standard protocol used throughout the duration of this project. This protocol was updated and revised after the 2013 field season to improve the accuracy and comparability of results from year-to-year and is described in Hand et al. (2016). One notable difference from this protocol is that we now perform two VES surveys within a 24-hour timeframe when possible to allow for estimating detection probabilities. The only exception in 2016 was Three Prong Lake, where we were only able to conduct one VES due to time constraints. The methods for statistical analysis conducted in 2016 are explained in detail in Hand et al. (2016), and Hand et al. (2018).

## RESULTS

In 2016, mountain lakes field personnel surveyed 24 lakes from four HUC5 watersheds. Fifteen of the 24 surveyed lakes contained fish; the other nine lakes were fishless. During visual encounter surveys, we detected CSF in 19 lakes and LTS in 6 lakes (Table 25).

### Fish surveys

Fifteen of the 24 lakes surveyed contained fish (Table 26). Seven of the lakes contained WCT, six contained BKT, and three contained RBT (Table 26). We sampled 378 fish total, 305 with gill nets and 73 with angling. Two of these lakes, Lake Creek lakes South and East were sampled twice for fish, in July and again in September. Of the 305 fish sampled via gill net, 160 were BKT, 99 were WCT, 41 were RBT, and 5 were WCT/RBT hybrids. Rod and reel samples were taken opportunistically and 73 fish, 44 WCT and 29 BKT were sampled in this manner. Gill net catch-per-unit-effort (CPUE; fish/h) ranged from 0.2 to 2.4/h, with an average of 1.15/h (Table 26). Angling CPUE varied between 1.0 and 31.7/h, but angling times varied from several minutes to several hours making a fair comparison difficult.

Due to shallow water, we were unable to set a gill net at Lookout Lake, however we were able to observe several RBT in the inlet, outlet, and lake itself. The fish observed in this “lake” are able to freely swim between Dan and Dodge lakes via streams connecting them and are expected to overwinter at one or the other to avoid perishing once Lookout Lake freezes over.

Lake Creek Lake West was sampled with a gill net, but no fish were collected. This was also the case the last time this lake was sampled in 2014. In neither case were fish observed by any field crew members. In 2016, we also visited the lake again later in the fall and it still was seemingly void of fish presence.

The relationship of length vs. weight of the major fish species that inhabited each lake this season as well as previous seasons is shown in Figure 103. These relationships have remained similar throughout the study.

**Amphibian abundance and distribution**

Columbia spotted frogs were detected in 19 of the 24 lakes surveyed in 2016 (79.2%), while LTS were found in six (25%; Table 27). Eleven (45.8%) of these lakes contained introduced trout species and CSF, three (12.5%) lakes contained both CSF and LTS, four (16.7%) lakes contained only CSF only, one lake (4.2%) contained only LTS, and three (12.5%) lakes contained all three. Columbia spotted frogs were also found to cohabitate with IGS at the only study site in which they were found, Three Prong Lake.

In the first and second rounds of sampling, CSF were detected at 63 lakes (85.1%) during both rounds, while LTS were found at 27 lakes (36.5%) in Round 1 and 37 lakes (50.0%) in Round 2 (Figure 104). In the first round 23 lakes (31.1%) contained fish along with CSF, while 24 lakes (32.4%) did in the second round. Four lakes (5.4%) in the first round contained both fish and LTS, which rose to seven lakes (9.5%) in the second round. Three of the lakes (4.1%) in the first round and six lakes (8.1%) in the second round contained fish, CSF, and LTS.

The third round is currently still underway with 63 of 74 (85.1%) lakes having been sampled for a third time. With the conclusion of the 2016 field season 50 of the 63 of the sampled lakes (79.4%) contained CSF, while 23 (36.5%) have LTS present, and 22 (34.9%) had both species present. In addition, 21 (33.3%) of the lakes comprised of both fish and CSF, but only four (6.3%) included CSF, LTS, and fish (Figure 104).

In 2016, seasonal trends (Julian Day and Julian Day$^2$) were found to be significant ($P$ <0.001) in regard to adult CSF occurrence. As in previous years we altered the binary response variable in the model so that it only treated lakes as having CSF present when there were at least three adult CSF recorded during the survey (CSF>2), which allowed us to investigate possibly overlooked variables that may be affecting CSF occurrence. After altering the binary response variable, we found that four explanatory variables were significant: the amount of Fines ($P$ = 0.048), Lake Depth ($P$ = 0.046), Julian Day ($P$ = 0.001), and Julian Day$^2$ ($P$ = 0.001). Fish Presence was not found to affect CSF occurrence with either response variable. When analyzing CSF abundance, we found that not only were the amount of Fines ($P$ = 0.006), Julian Day ($P$ <0.001), and Julian Day$^2$ ($P$ <0.001) significant as in the occurrence model, but also Fish Presence ($P$ <0.001) and Snow Depth ($P$ <0.001) that spring.

Unlike CSF, our analysis of LTS occurrence found that Fish Presence ($P$ <0.001) in addition to Julian Day ($P$ <0.001) and Julian Day$^2$ ($P$ <0.001) to be significant. The abundance of LTS was found to be affected by six significant variables: Lake Depth ($P$ = 0.05), Fish Presence ($P$ < 0.001), Julian Day ($P$ = 0.003), Julian Day$^2$ ($P$ <0.001), Snow Depth ($P$ <0.001), and the amount of Fines ($P$ <0.001).

**Garter Snake abundance and distribution**

Garter snakes were detected at 50% of the lakes (12 of 24) sampled this past season. We were able to mirror the seasonal and habitat models used for the amphibian occurrence and abundance analyses to model GS occurrence. Four variables predicted GS occurrence including: Lake Elevation ($P$ = 0.004), amount of Fines ($P$ = 0.03), Julian Day$^2$ ($P$ = 0.01), and Snow Depth ($P$ = 0.04). In addition to Lake Elevation ($P$ = 0.006), the variables Fines ($P$ = 0.009), Julian Day$^2$ ($P$ = 0.01), and Snow Depth ($P$ <0.001) also predicted Fish Presence ($P$ <0.001).

**Long-term trends in presence and abundance**

There does not seem to be a significant long-term trend in amphibian presence (number of lakes occupied. However, there were significant positive trends for both CSF ($P$ <0.001) and LTS ($P$ <0.001) abundance in regard to Year, meaning that abundance has been increasing through time.

**Seasonal variation in amphibian presence**

During the 2016 field season, we were able to sample Bleak Creek, Goat, and the Lake Creek Lakes (South, West, and East) in the Bargamin Creek drainage twice. This was done previously in 2015 on a different set of lakes in order to gain a better understanding of seasonal variation in amphibian presence or detection. The first survey took place July 20-23, 2016 and the second September 14-17, 2016. During the July survey, CSF were present in four of the five lakes (80%), and LTS were present in three of the five lakes (60%). During the September survey, CSF were present in four of the five lakes (80%), and LTS were present in three of the five lakes (60%). While LTS were present in the same number of lakes for both the July and September trips, we were unable to detect them in Bleak Creek Lake during the September survey, where they had been previously found during the July survey and vice-versa for Lake Creek Lake South.

## DISCUSSION

**Fish surveys**

During the 2016 field season, we sampled 378 fish from 15 lakes. This season marked the third time the majority of these lakes were surveyed excluding MacArthur, Stillman, and Boston lakes, which were surveyed for the fourth time. The numbers of fish sampled in the first three rounds varies with the first round resulting in 392 fish, the second round with 293 fish, and the third round with 311 fish (Table 28). The fourth round for this subset of lakes has yet to be completed.

Although we set a gill net on Lake Creek Lake, we did not sample any fish. This was also the case when the lake was previously sampled in 2014. In neither case did any members of the field crew observe fish activity, in 2016 we also visited the lake again later in the fall and it still was seemingly void of fish. Thus, it is in my opinion that Lake Creek Lake West should now be considered a fishless.

The length vs. weight relationships of major fish species have remained similar throughout the study (Figure 103). However, it is important to consider that stocking occurs in only three lakes, and sampling practices have not been consistent throughout this study. These factors may affect the fish population parameters such as average length, weight, age, or density.

## Habitat variables

Habitat relationships for both LTS and CSF were generally consistent with previous studies (Pilliod et al. 1996; Murphy 2002). As found in previous years, CSF occurrence was positively correlated to the amount of Fines and Depth of the lake; however, as was stated in previous years' reports lake depth is auto correlated with lake perimeter and thus should be interpreted with caution. The abundance model on the other hand accounts for this possible bias and offsets the lake perimeter. We also found that the presence of Fish did not seem to have a significant effect on CSF occurrence within the study.

Fish do significantly affect both LTS occurrence and abundance. This is likely attributable to the longer larval stage of LTS (relative to CSF) which increases the susceptibility to predation during this life stage. The abundance of LTS was also significantly affected by the depth of the Snowpack that spring as well as the amount of Fines present in the lake. However, the presence of many fishless lakes allows for widespread occurrence and meta-population stability. This is substantiated by our results indicating no long-term trends in abundance.

While this year's analysis supports the analyses of previous years there are several variables that are important to acknowledge. Over the years, there have been shifts in survey protocol and criteria. Most recently, we have simplified the stratification of life stages from adult, sub-adult, larvae, and egg mass to remove the category of sub-adult in order to improve surveyor consistency. Until recently, only one VES was conducted at each lake, but this was increased to two surveys once it was discovered that an additional VES would improve detection rates, especially for LTS.

## Temporal variables

As was seen in previous years, Julian Day and Julian Day$^2$ proved to be highly significant variables in every count and occurrence model. This strong seasonality should be taken into account in future analyses, as ignoring it can lead to erroneous interpretations. In addition, since lakes are not always surveyed during the same day, week, or even month during each round of surveys it may make comparing the occurrence and abundance for a given lake from year to year inaccurate. Based on our analysis, July-August are the best months for sampling amphibians. This is the time of year when they are most active, and therefore most visible.

## Long-term trends

As predicted by the power analyses conducted in previous years, we detected no significant trends in occupancy of either CSF or LTS with the addition of the 2016 data. The abundance models indicate significant positive trends, but as was stated in 2014, these results should be interpreted with caution since the data collected in 2016 composes a significant portion of the surveys in the current data set. However, through time a larger dataset will help to mitigate the biases of any one season and give a clearer picture of long-term population trends.

## Garter Snakes

The abundance of GS was found to be significantly impacted by the presence of fish, which is likely due to the fact that the presence of fish reduces GS preferred prey, amphibians. This finding warrants further analysis and investigation. In future analysis, we should also look at the potential correlation between GS and amphibian occurrence and abundance. A study conducted in the Sierra Nevada Mountains indicated that the probability of finding GS was 30 times higher at lakes with amphibians versus lakes without (Matthews et al. 2002). Elevation was found to be significant for both occurrence and abundance of GS, which may be attributed to several variables such as nutrient poor environments or thermal barriers.

## MANAGEMENT RECOMMENDATIONS

1. Continue monitoring high mountain lakes within HUC5 watersheds in the Clearwater Region as part of the long-term amphibian risk assessment.

2. As smaller lentic areas dry or infill, lake number and surface area reduction should be updated to determine if HUC5 watersheds change in amphibian risk classification. Ex. Eagle Creek is still considered a site, yet was dry in 2010 and 2016, and has quite possibly been dry prior to 2010.

3. Analyze amphibians on a population based scale rather than in terms of presence/absence to provide a more precise measure of population trends.

4. Continue to sample a specific set of lakes twice a year, once in the early season and once in the late season to improve our understanding of how seasonality affects amphibian populations.

5. Continue conducting multiple VES during one visit to each lake surveyed in a season to improve LTS detection probabilities.


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Table 25. Columbia Spotted Frog (CSF), Long-toed Salamander (LTS), and fish presence during high mountain lake surveys in the Clearwater Region, 2016.

| | | | | | Presence | | |
|---|---|---|---|---|---|---|---|
| Lake | Risk | HUC5 | HUC4 | Survey date | Fish | CSF | LTS |
| MacArthur | Elevated | Bargamin Cr. | Middle Salmon | 7/7/2016 | Yes | Yes | No |
| Three Prong* | Elevated | Bargamin Cr. | Middle Salmon | 7/8/2016 | No | Yes | No |
| Eagle Creek** | Moderate | Running Cr. | Lochsa | 7/9/2016 | No | No | No |
| Stillman | Elevated | Bargamin Cr. | Middle Salmon | 7/9/2016 | Yes | Yes | No |
| Bleak | Elevated | Bargamin Cr. | Middle Salmon | 7/20/2016 | No | Yes | Yes |
| Lake Cr. South | Elevated | Bargamin Cr. | Middle Salmon | 7/21/2016 | Yes | Yes | No |
| Lake Cr. West | Elevated | Bargamin Cr. | Middle Salmon | 7/22/2016 | No | Yes | No |
| Lake Cr. East | Elevated | Bargamin Cr. | Middle Salmon | 7/22/2016 | Yes | Yes | Yes |
| Goat BC | Elevated | Bargamin Cr. | Middle Salmon | 7/23/2016 | No | No | Yes |
| Boston Mountain | Elevated | Bargamin Cr. | Middle Salmon | 7/25/2016 | Yes | Yes | Yes |
| Florence | Elevated | Old Man Cr. | Lochsa | 8/5/2016 | Yes | Yes | No |
| Hjort | Elevated | Old Man Cr. | Lochsa | 8/5/2016 | Yes | Yes | No |
| Elizabeth | Elevated | Old Man Cr. | Lochsa | 8/6/2016 | Yes | Yes | No |
| Lloyd | Elevated | Old Man Cr. | Lochsa | 8/7/2016 | Yes | No | No |
| Dishpan | Elevated | Old Man Cr. | Lochsa | 8/7/2016 | Yes | Yes | No |
| Flea | Elevated | Old Man Cr. | Lochsa | 8/18/2016 | No | Yes | Yes |
| Chimney | Elevated | Old Man Cr. | Lochsa | 8/19/2016 | Yes | Yes | No |
| Old Man | Elevated | Old Man Cr. | Lochsa | 8/20/2016 | Yes | Yes | No |
| Kettle | Elevated | Old Man Cr. | Lochsa | 8/21/2016 | No | Yes | No |
| Wood | Elevated | Old Man Cr. | Lochsa | 8/22/2016 | No | Yes | Yes |
| Dan | Low | Storm Cr. | Lochsa | 9/2/2016 | Yes | No | No |
| Lookout | Low | Storm Cr. | Lochsa | 9/2/2016 | Yes | Yes | No |
| Dodge SC | Low | Storm Cr. | Lochsa | 9/2/2016 | Yes | Yes | No |
| Maud | Low | Storm Cr. | Lochsa | 9/3/2016 | No | Yes | No |
| Bleak | Elevated | Bargamin Cr. | Middle Salmon | 9/14/2016 | No | Yes | Yes |
| Lake Cr. South | Elevated | Bargamin Cr. | Middle Salmon | 9/15/2016 | Yes | Yes | Yes |
| Lake Cr. West | Elevated | Bargamin Cr. | Middle Salmon | 9/15/2016 | No | Yes | No |
| Lake Cr. East | Elevated | Bargamin Cr. | Middle Salmon | 9/15/2016 | Yes | Yes | No |
| Goat BC | Elevated | Bargamin Cr. | Middle Salmon | 9/16/2016 | No | No | Yes |

* Idaho Giant Salamander larvae present.

**Lake was dry.

Table 26. Summary of catch-per-unit-effort (CPUE), average total length, and weight of fish captured during high mountain lake surveys in the Clearwater Region, 2016.

| Lake | Species | CPUE (Gillnet) | Average length (mm) | Average weight (g) |
|---|---|---|---|---|
| MacArthur | WCT | 0.33 | 213 | 113 |
| Stillman | WCT | 0.79 | 174 | 64 |
| Lake Cr. South | WCT | 0.19 | 285 | 243 |
| Lake Cr. East | WCT, WCTxRBT | 1.7 | 202 | 110 |
| Boston Mountain | WCT | 1.1 | 153 | 41 |
| Florence | WCT | 0.29 | 313 | 388 |
| Hjort | WCT, BKT | 1.45 | 150 | 38 |
| Elizabeth | BKT | 2.1 | 168 | 48 |
| Lloyd | BKT | 2.43 | 187 | 73 |
| Dishpan | BKT | 0.62 | 187 | 75 |
| Chimney | BKT | 0.74 | 201 | 88 |
| Old Man | BKT | 1.33 | 216 | 107 |
| Dan | RBT | 1.7 | 211 | 94 |
| Dodge SC | RBT | 1.3 | 249 | 191 |

WCT = Westslope Cutthroat Trout
RBT = Rainbow Trout
BKT = Brook Trout

Table 27. Fish and amphibian presence in Clearwater Region high mountain lakes.

| | | | Historical | | First round | | Second round | | Third round | |
|---|---|---|---|---|---|---|---|---|---|---|
| Lake | Huc5 | Risk | Fish | Amphibians | Fish | Amphibians | Fish | Amphibians | Fish | Amphibians |
| Bilk Mountain | Goat Creek | Control | NONE | CSF | NONE | CSF/ LTS | NONE | CSF | NONE | NONE |
| Goat | Goat Creek | Control | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS | -- | -- |
| Mud | Goat Creek | Control | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | -- | -- |
| Bilk | Upper Meadow | Control | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS | -- | -- |
| Elk | Upper Meadow | Control | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Section 27 | Upper Meadow | Control | -- | -- | NONE | CSF/ LTS | NONE | LTS | -- | -- |
| Big Harrington #1 | Big Harrington | Control | -- | -- | NONE | NONE | NONE | NONE | -- | -- |
| Big Harrington #6 | Big Harrington | Control | -- | -- | NONE | CSF | NONE | NONE | -- | -- |
| Fox Peak Lower | NF Moose Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Fox Peak Upper | NF Moose Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Isaac Creek | NF Moose Creek | Low | -- | -- | NONE | CSF | NONE | CSF/ LTS | NONE | CSF |
| Isaac | NF Moose Creek | Low | WCT/ RBT | CSF | WCT/ RBT | CSF | WCT | CSF | WCT | CSF |
| Section 28 | NF Moose Creek | Low | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | -- | -- |
| West Moose #1 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| West Moose #2 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | -- | -- |
| West Moose #3 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| West Moose #4 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| West Moose #5 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS |
| West Moose #6 | NF Moose Creek | Low | -- | -- | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| West Moose #7 | NF Moose Creek | Low | -- | -- | NONE | CSF | NONE | CSF | NONE | CSF/ LTS |
| West Moose #8 | NF Moose Creek | Low | -- | -- | NONE | CSF | NONE | LTS | NONE | CSF |
| West Moose #9 | NF Moose Creek | Low | -- | -- | NONE | CSF | NONE | CSF | NONE | CSF |
| Dan | Storm Creek | Low | RBT | CSF | RBT | CSF | RBT | CSF | RBT | NONE |
| Dodge | Storm Creek | Low | RBT | CSF | RBT | CSF | RBT | CSF | RBT | NONE |
| Lookout | Storm Creek | Low | RBT | CSF | RBT | CSF | RBT | CSF | RBT | CSF |
| Maud | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF | NONE | CSF |
| Middle Storm | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF |
| North Sec. 25 | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| North Storm | Storm Creek | Low | NONE | CSF | NONE | CSF | NONE | CSF/ LTS | NONE | NONE |
| N.E. Ranger | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Old Stormy | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF | -- | -- |
| Ranger | Storm Creek | Low | RBT | CSF | RBT | NONE | RBT | CSF/ LTS | RBT | NONE |
| Section 27 | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Siah | Storm Creek | Low | WCT/ RBT | CSF | WCT/ RBT | CSF | WCT/RBT | CSF/ LTS | WCT | CSF/ LTS |
| South Sec. 25 | Storm Creek | Low | NONE | CSF/ LTS | NONE | CSF | NONE | CSF | NONE | CSF/ LTS |
| Storm | Storm Creek | Low | NONE | CSF/ LTS | NONE | NONE | NONE | LTS | NONE | NONE |

Abbreviation Key: CSF = Columbia Spotted Frog, LTS = Long-toed Salamander, TF = Rocky Mountain Tailed Frog, IGS = Idaho Giant Salamander, WCT = Westslope Cutthroat Trout, RBT = Rainbow Trout, BKT = Brook Trout.

Table 27. (continued)

| Lake | Huc5 | Risk | Historical | | First round | | Second round | | Third round | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Fish | Amphibians | Fish | Amphibians | Fish | Amphibians | Fish | Amphibians |
| Eagle Creek* | Running Creek | Moderate | -- | -- | NONE | NONE | NONE | NONE | NONE | NONE |
| Running | Running Creek | Moderate | BKT | CSF | BKT | NONE | BKT | CSF | BKT | CSF |
| Section 26 Lower | Running Creek | Moderate | -- | -- | NONE | NONE | NONE | CSF | NONE | NONE |
| Section 26 Upper | Running Creek | Moderate | -- | -- | NONE | LTS | NONE | NONE | NONE | NONE |
| Dodge | Warm Springs Crk. | Moderate | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | -- | -- |
| East Wind | Warm Springs Crk. | Moderate | WCT | CSF/ LTS | WCT | CSF | NONE | CSF | WCT | CSF |
| Hungry | Warm Springs Crk. | Moderate | WCT/ RBT | CSF | WCT | CSF | WCT | CSF | -- | -- |
| Low . N. Wind | Warm Springs Crk. | Moderate | NONE | CSF/ LTS | NONE | NONE | NONE | NONE | NONE | NONE |
| Middle Wind | Warm Springs Crk. | Moderate | WCT | CSF | WCT | CSF | WCT | CSF | WCT | CSF |
| N.W. Wind | Warm Springs | Moderate | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF | NONE | CSF |
| South Wind | Warm Springs Crk. | Moderate | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Up. N. Wind | Warm Springs Crk. | Moderate | NONE | LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| West Wind | Warm Springs Crk. | Moderate | WCT | CSF | WCT | CSF | WCT | CSF/ LTS | WCT | CSF/ LTS |
| Wind Pond | Warm Springs Crk. | Moderate | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Bleak Creek | Bargamin Creek | Elevated | NONE | CSF/ LTS | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS |
| Boston Mtn. | Bargamin Creek | Elevated | WCT | CSF/ LTS | WCT | CSF | WCT | CSF | WCT | CSF |
| Goat Lake | Bargamin Creek | Elevated | WCT | LTS | NONE | LTS | NONE | LTS | NONE | LTS |
| Lake Creek E. | Bargamin Creek | Elevated | WCT/ RBT/X | CSF | WCT/ RBT/X | CSF/ LTS | WCT | CSF | WCT/ RBT/X | CSF/ LTS |
| Lake Creek. S. | Bargamin Creek | Elevated | WCT/ RBT | CSF | RBT | CSF/TF | NONE | CSF | WCT | CSF/ LTS |
| Lake Creek W. | Bargamin Creek | Elevated | RBT | CSF | RBT | CSF | WCT | CSF | NONE | CSF |
| MacArther | Bargamin Creek | Elevated | WCT/ RBT | CSF/ LTS | WCT/ RBT | CSF | WCT/RBT | CSF/ LTS | WCT | CSF |
| Stillman | Bargamin Creek | Elevated | WCT | CSF | WCT | CSF/ LTS | WCT | CSF/ LTS | WCT | CSF |
| Three Prong | Bargamin Creek | Elevated | -- | -- | NONE | CSF/ IGS | NONE | CSF/ IGS | NONE | CSF/ IGS |
| Chimney | Old Man Creek | Elevated | BKT | NONE | BKT | CSF | BKT | CSF | BKT | CSF |
| Dishpan | Old Man Creek | Elevated | BKT | CSF | BKT | CSF | BKT | CSF | BKT | CSF |
| Elizabeth | Old Man Creek | Elevated | BKT/ WCT | CSF | BKT/ WCT | NONE | BKT/WCT | NONE | BKT | NONE |
| Flea | Old Man Creek | Elevated | NONE | CSF | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF |
| Florence | Old Man Creek | Elevated | WCT | CSF/ LTS | WCT | CSF/ LTS | WCT | CSF | WCT | CSF |
| Hjort | Old Man Creek | Elevated | BKT | CSF | BKT | CSF | BKT/WCT | CSF | BKT/ WCT | CSF |
| Kettle | Old Man Creek | Elevated | RBT | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF |
| Lloyd | Old Man Creek | Elevated | BKT | NONE | BKT | NONE | BKT | NONE | BKT | NONE |
| Lottie | Old Man Creek | Elevated | -- | -- | BKT | CSF | BKT | CSF | BKT | CSF |
| Lottie Upper | Old Man Creek | Elevated | BKT | CSF | BKT | CSF | BKT | CSF | BKT | CSF |
| Maude East | Old Man Creek | Elevated | RBT | CSF | RBT | CSF | WCT/HY | CSF/ LTS | WCT | CSF |
| Maude North | Old Man Creek | Elevated | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF | NONE | CSF |
| Maude West | Old Man Creek | Elevated | RBT | CSF | RBT | CSF | WCT/HY | CSF/ LTS | WCT | CSF |
| Old Man | Old Man Creek | Elevated | BKT | CSF | BKT | CSF | BKT | CSF | BKT | CSF |
| Wood | Old Man Creek | Elevated | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS | NONE | CSF/ LTS |

Abbreviation Key: CSF = Columbia Spotted Frog, LTS = Long-toed Salamander, TF = Rocky Mountain Tailed Frog, IGS = Idaho Giant Salamander, WCT = Westslope Cutthroat Trout, RBT = Rainbow Trout, BKT = Brook Trout.

Table 28. Number of fish sampled at each mountain lake by gill nets during each round of sampling. MD = missing data.

| | Round | | | |
|---|---|---|---|---|
| Lake | 1 | 2 | 3 | 4 |
| MacArthur | 27 | 24 | 5 | 4 |
| Stillman | 21 | 24 | 8 | 11 |
| Lake Creek South | 8 | 24 | 10 | --- |
| Lake Creek East | 31 | 16 | 29 | --- |
| Boston | 32 | 21 | 20 | 33 |
| Florence | md | 15 | 8 | --- |
| Hjort | 28 | 5 | 48 | --- |
| Elizabeth | 33 | 44 | 43 | --- |
| Lloyd | 73 | md | 51 | --- |
| Dishpan | 20 | md | 13 | --- |
| Chimney | 43 | 51 | 18 | --- |
| Old Man | 27 | 28 | 18 | --- |
| Dan | 26 | 24 | 20 | --- |
| Dodge | 23 | 17 | 20 | --- |
| Totals | 392 | 293 | 311 | 48 |

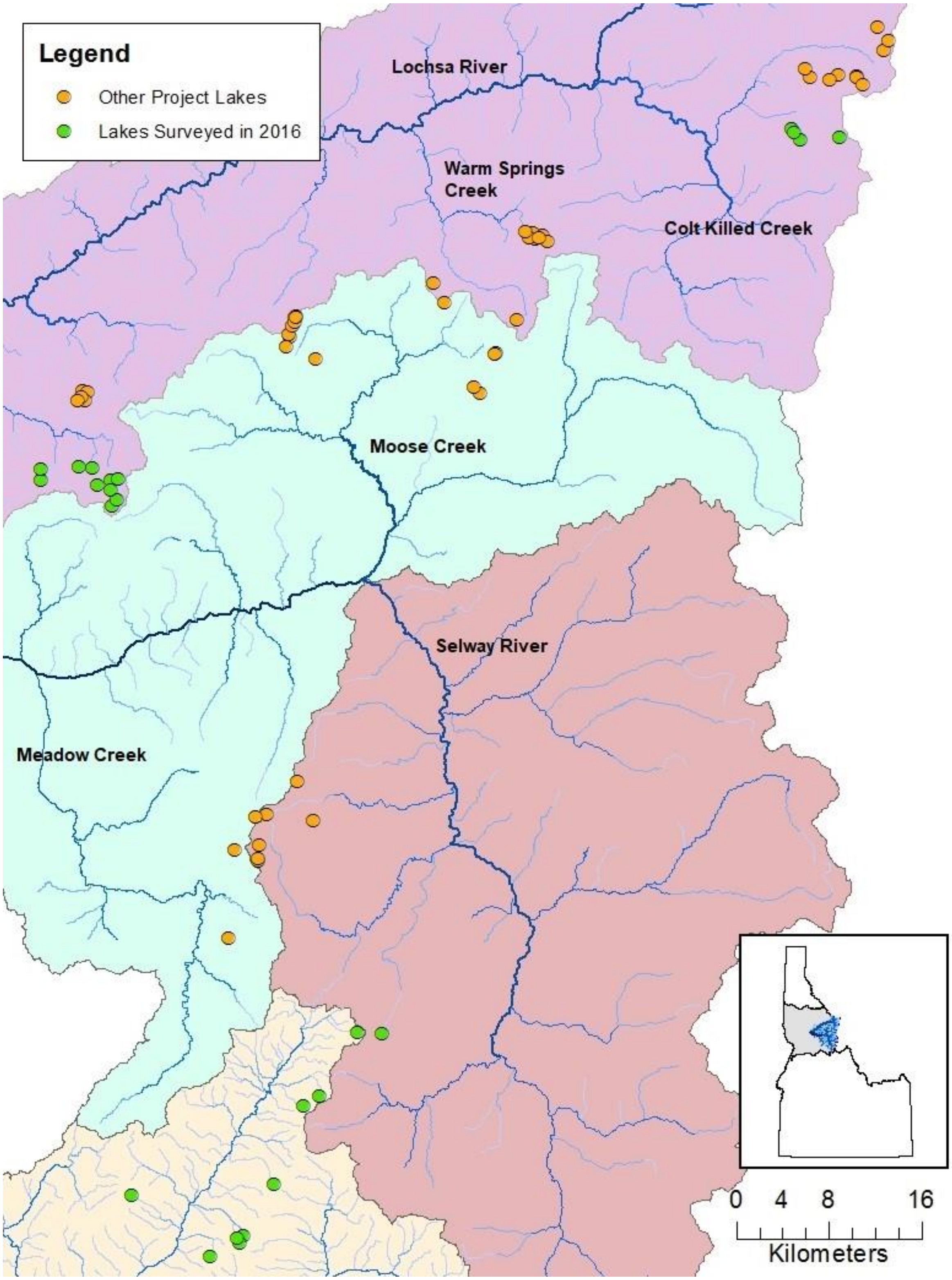



Figure 102. Map of high mountain lakes surveyed in the Clearwater Region of Idaho, during 2016.

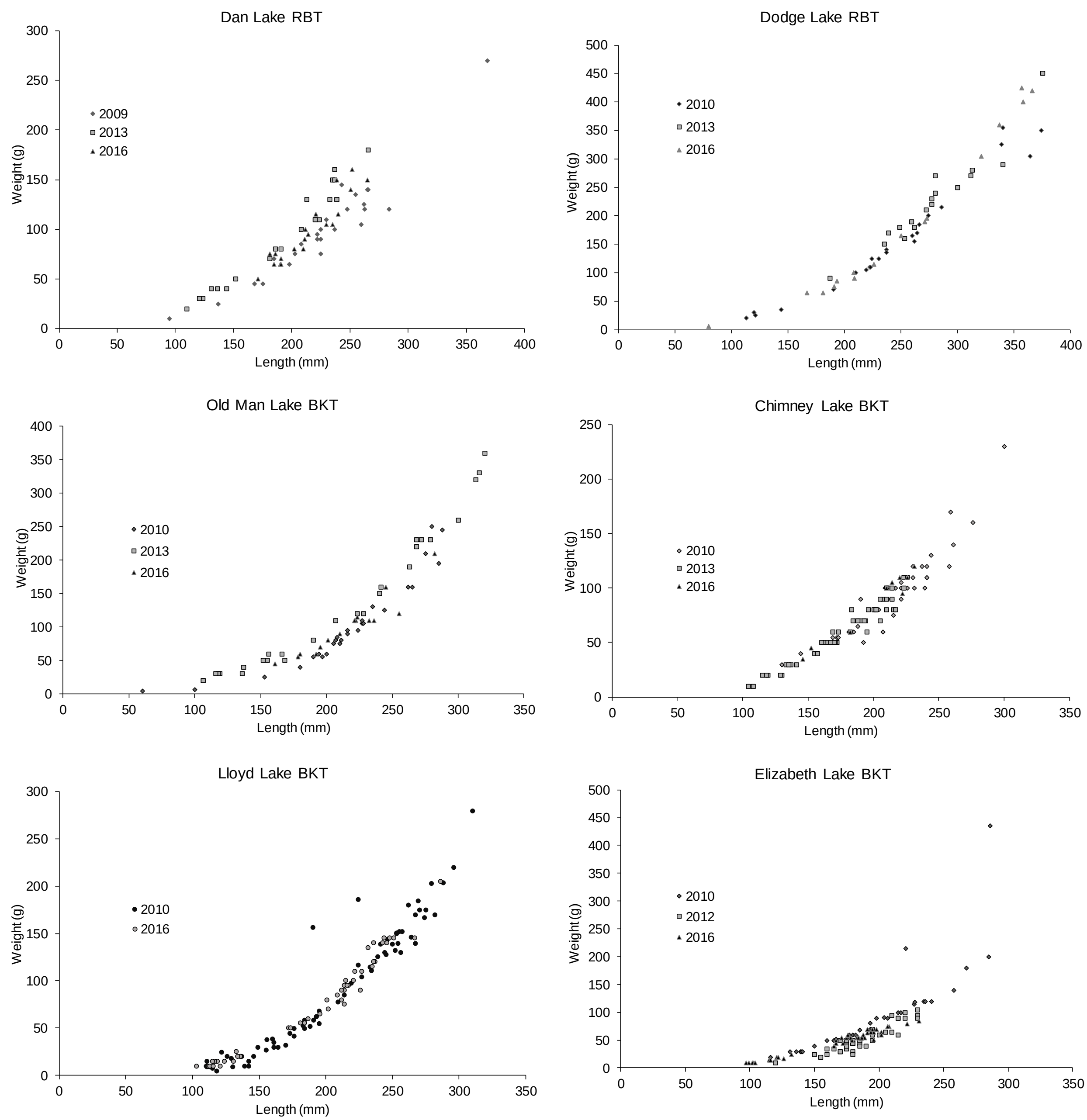
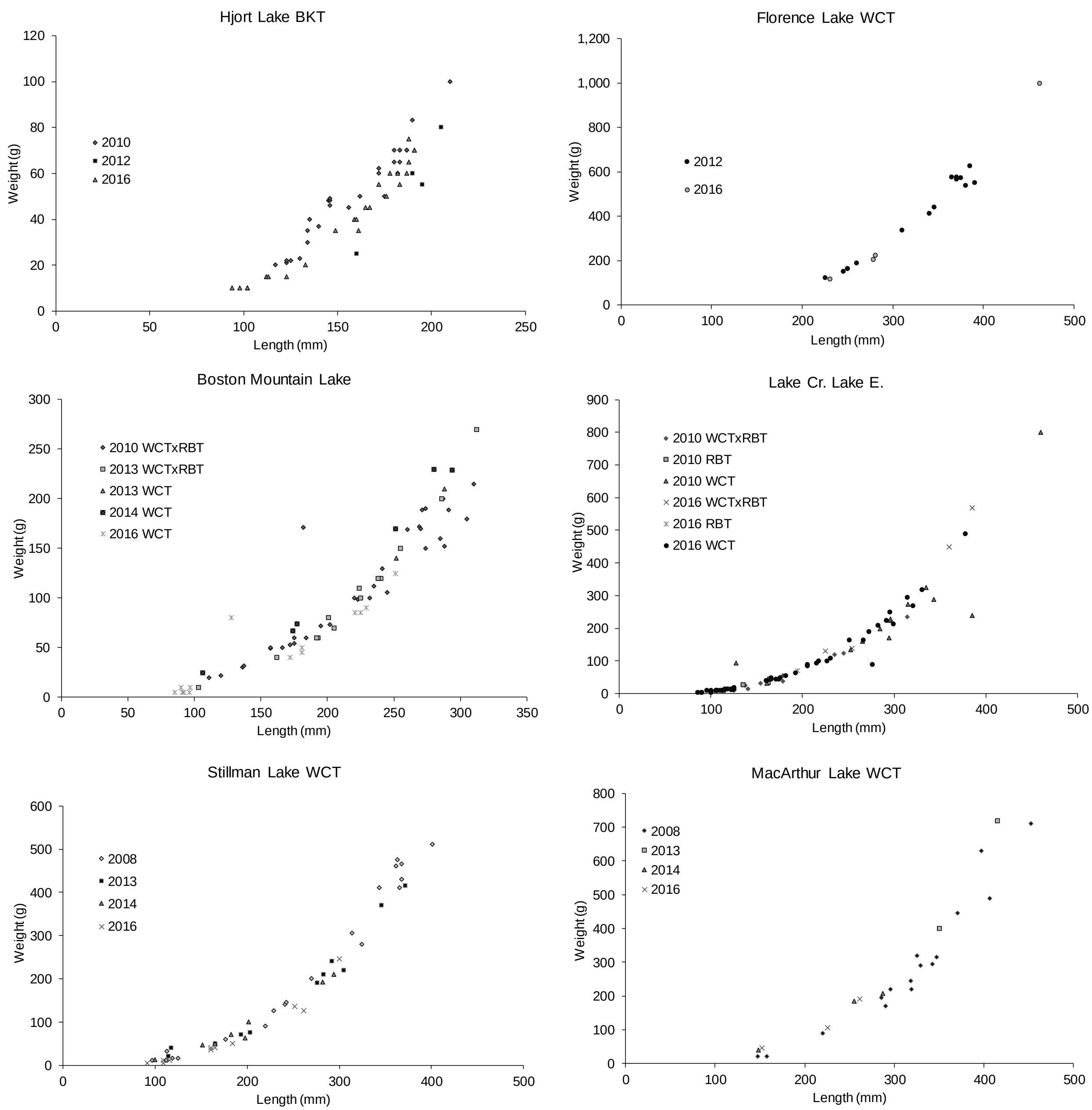



Figure 103. Length vs. weight distributions of trout caught by gill net in 2016, from high mountain lakes in the Clearwater Region compared to previous gillnetting efforts.

Figure 103. (continued)



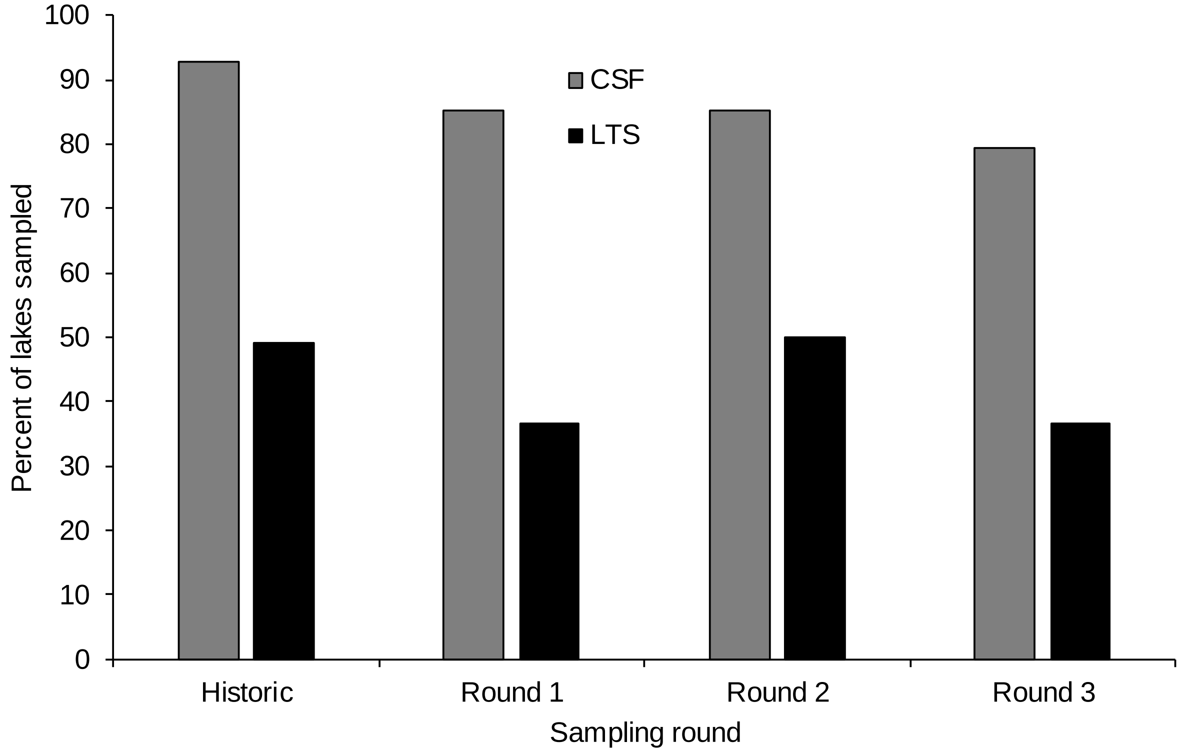



Figure 104. Presence of Columbia Spotted Frogs (CSF) and Long-toed Salamanders (LTS) during visual encounter surveys of high mountain lakes of the Clearwater Region, Idaho.

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